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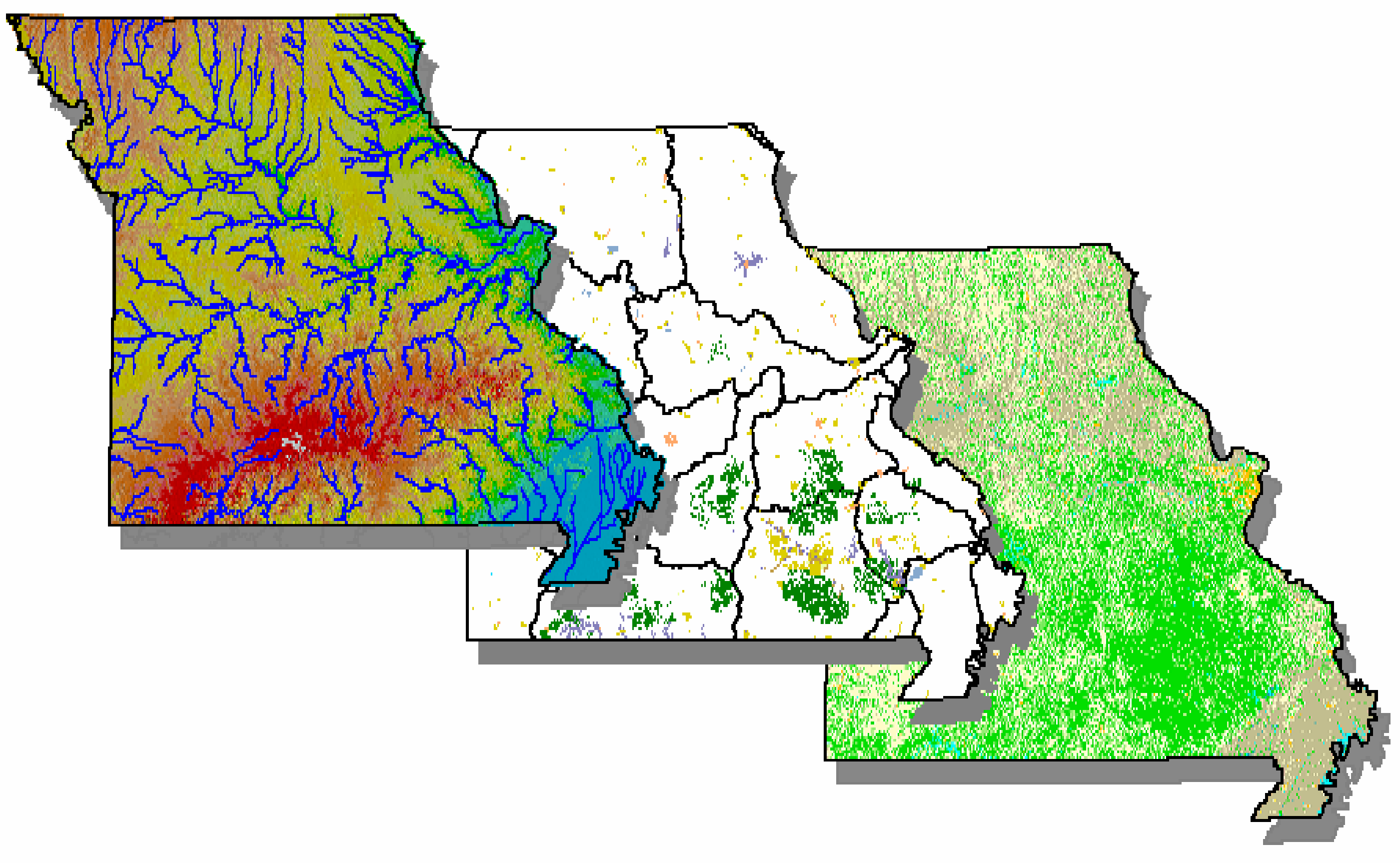
A GAP ANALYSIS FOR RIVERINE ECOSYSTEMS OF MISSOURI

National Gap Analysis Program

2005 Final Report



A GEOGRAPHIC APPROACH TO PLANNING FOR BIOLOGICAL DIVERSITY

U.S. Department of the Interior
U.S. Geological Survey

A GAP ANALYSIS FOR RIVERINE ECOSYSTEMS OF MISSOURI

FINAL REPORT

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EXECUTIVE SUMMARY

The National Gap Analysis Program (GAP) was initiated in 1988 to provide a coarse-filter assessment strategy for identifying and prioritizing biodiversity conservation needs. While GAP has made enormous strides in developing and conducting coarse-filter biodiversity assessments for terrestrial ecosystems, much less has been accomplished for aquatic ecosystems. The need for developing an aquatic component of GAP was recognized as early as 1993, when Congress allocated the funds needed to support such an effort. Those funds, however, were rescinded. GAP did manage to initiate an aquatic component of the program in 1995 with a pilot in the upper Allegheny River Basin in Western New York. In 1997, in cooperation with the Missouri Resource Assessment Partnership (MoRAP) and financial assistance by the USGS National Water Quality Assessment Program, the U.S. Department of Defense-Legacy Program, and the Missouri Department of Conservation, GAP initiated a statewide pilot project for the state of Missouri. Both of these projects focused on riverine ecosystems. This report summarizes the approach, results and significant findings of the Missouri pilot project.

When it comes to freshwater ecosystems the North American continent, and in particular the United States, harbors an astounding proportion of the world's freshwater species. Despite this distinction, North America and the United States are facing a freshwater biodiversity crisis. While much attention has been focused on the global losses of terrestrial biodiversity especially in tropical ecosystems, comparatively little attention has been given to the alarming declines in freshwater biodiversity. Yet, it is encouraging to see that within the last decade more and more attention has been focused on conserving freshwater biodiversity. A critical first step to slowing the loss of biodiversity is identifying gaps in existing efforts to conserve freshwater biodiversity across the landscape and then prioritizing opportunities to fill these gaps. This is the overall goal of the USGS National Gap Analysis Program and this project.

The principal goal of our project was to identify riverine ecosystems and species not adequately represented (i.e., gaps) in the matrix of conservation lands in Missouri. Another goal was to develop ways of integrating the terrestrial and aquatic components of gap analysis. In addition, we wanted to provide spatially explicit data that could be used by natural resource professionals, legislators, and the public to make more informed decisions for prioritizing opportunities to fill these conservation gaps and to devise strategic approaches for developing effective long-term biodiversity conservation plans. Furthermore, as a pilot project for a national program, we also had the goal of developing a broadly applicable gap analysis methodology. We addressed this goal by ensuring that we utilized nationally standardized and available geospatial data wherever possible and also by devising a flexible conservation assessment methodology, which can accommodate the differences in data availability (e.g., biological) that exists among states across the United States.

Several geospatial and tabular datasets were developed to meet the information/data needs for identifying conservation gaps and subsequently prioritizing opportunities to

fills these gaps: a) maps of a hierarchical classification of riverine ecosystems, b) predicted species distribution maps, c) ownership and stewardship maps, and d) maps of human stressors. These data were then used to conduct a gap analysis of both biotic and abiotic conservation targets and also to develop a statewide freshwater biodiversity conservation plan.

The data and methods developed and used in this project go well beyond anything done to date in any part of the world. Our assessment methods incorporated both ecological and evolutionary contexts that are so critical to conserving biodiversity, which heretofore have been largely ignored. Also, the high resolution biological and stewardship data (i.e., individual stream segment) coupled with the tremendous amount of geospatial data on human stressors enabled us to precisely pinpoint specific areas (clusters of stream segments) that are critical to the long-term maintenance of biodiversity within Missouri.

Even though the basic goal and objectives of the terrestrial and aquatic components of gap are the same, there is a major obstacle to upfront integration of the gap analyses. The foremost obstacle to a fully integrated terrestrial and aquatic gap analysis pertains to the fact that if we are going to conserve biodiversity we must conserve ecosystems. Traditionally, ecoregions have served as the geographic framework for defining terrestrial ecosystems and conserving terrestrial biodiversity. While ecoregions do a good job of accounting for structural and functional differences in freshwater ecosystems, they do not account for important compositional differences (species and genetic composition) that have resulted from isolation mechanisms largely related to historical and contemporary drainage patterns. Defining ecosystems in freshwater environments requires the integration of ecoregion and watershed boundaries. Consequently, separate geographic frameworks are needed in order to appropriately place terrestrial and aquatic ecosystems into their proper ecological and evolutionary contexts. This is why we developed a separate aquatic ecological classification framework for our project. This fundamental difference should not be viewed as an impediment to conserving biodiversity, merely an obstacle. Separate conservation assessments or gap analyses can be performed and the results then integrated a posteriori into an overall assessment or analysis. This is the approach we have taken in Missouri.

The results of the gap analysis are not encouraging. However, the results from the conservation planning efforts provide hope that relatively intact ecosystems still exist even in highly degraded landscapes. Results also suggest that a wide spectrum of the abiotic and biotic diversity can be represented within a relatively small portion of the total resource base, with the understanding that for riverine ecosystems the area of conservation concern is often substantially larger than the identified priority locations. Selecting and mapping priority riverscapes for conservation is the first step toward effective biodiversity conservation. Yet, establishing geographic priorities is only one of the many steps in the overall process of achieving real conservation. Achieving the ultimate goal of conserving biodiversity will require vigilance on the part of all responsible parties, with particular attention to addressing and coordinating the many remaining logistical tasks.

We have held nine training workshops in order to provide training to individuals interested in implementing our methods in their respective states. Through these training workshops we have provided training to more than 50 individuals representing numerous state and federal agencies and academic institutions. This training has helped with the establishment of aquatic gap projects in 20 states.

ACKNOWLEDGEMENTS

We first want to thank to John Mosesso and Doug Beard of the US Geological Survey (USGS) Biological Resource Division and Kevin Gergely of the USGS National Gap Analysis Program for their unwavering support of our project over the years. We also want to thank several past and present staff of the National Gap Analysis Program. Specifically, we want to thank Mike Jennings for selecting Missouri as one of the first pilot projects for the aquatic component of GAP and giving MoRAP the chance to take on this task. We also need to thank Mike and Patrick Crist for their invaluable advice and the many thoughtful theoretical and technical discussions during the early stages of the project. We are very grateful to Elizabeth Brackney, Ree Brannon, and Becky Sorbel for all of their administrative support over the years.

The USGS National Water Quality Assessment Program, the US Department of Defense Legacy Program, and the Missouri Department of Conservation (MDC) provided funding for this project. We want to thank Rachel Muir (USGS) for providing the initial financial support for our project and her theoretical and technical assistance during the early stages of the project. We are especially grateful to Leslie Orzetti for her support and administration of this project through the Legacy Program. Joe Dillard and Gary Novinger, of MDC, deserve special thanks for their foresight and fortitude in garnering support for this project throughout their agency and Missouri.

To say that this project was a collaborative effort would be a gross understatement. There are an overwhelming number of people who have generously contributed their time and effort to this project and without whom this project could never have been completed. We sincerely regret any omissions, because everyone who has contributed deserves a special “Thank You”!

We are deeply indebted to all of the past and present staff at The Nature Conservancy’s Freshwater Initiative Program for their foundational work on developing a hierarchical ecological classification for freshwater ecosystems. We would especially like to thank Jonathan Higgins, Mary Lammert-Khoury, and Denny Grossman (NatureServe) for all of their open collaboration on this difficult task over the years. A special thanks to Mary and Jonathan, Paul Angermeier (Virginia Tech University), Paul Seelbach (Michigan DNR), and Randy Sarver (Missouri DNR) for all of their thought provoking discussions on this topic! Tom Fitzugh and Mark Caldwell (MDC) also deserve special thanks for their diligent work on overcoming the many technical hurdles involved with trying to efficiently map classification units within a GIS. Pamela Haverland (USGS), Jeri Peck (EcoStats), and John Stanovick (MDC, now US Forest Service) provided invaluable statistical assistance that was so critical to the classification process.

This project could not have been possible without the countless hours of fieldwork by those individuals over the past 100 years that have tirelessly gone into the wilds of Missouri to make the thousands of biological collections used as the basis for predictive models eventually developed in this project. We must also thank the many agencies and individuals that took the time to organize and share these collection records with us. Especially, the MDC, Missouri Department of Natural Resources, USGS, EPA Region 7, University of Missouri, and Ronald Oesch. Special thanks go out to those individuals that took the time to review the many distribution maps created during this project; Sue Brundermann, Alan Buchanan, Bob DiStefano, Scott Faiman, Ronald Oesch, and William Pflieger. Dr. Pflieger deserves special recognition for sharing his tremendous knowledge of Missouri's stream resources with us. Pamela Haverland and John Stanovick again deserve special thanks for their statistical and database management assistance, which saved countless months of work on this aspect of the project. Dorothy Butler (MDC), thanks for your assistance with the Missouri Heritage Database. Ted Hoehn (Florida Fish and Wildlife Commission) provided valuable advice on developing range maps for freshwater species. James Peterson (GA Cooperative Fish and Wildlife Research Unit) deserves thanks for his assistance on the nuisances of the many statistical approaches to modeling species distributions. Thanks also to Julie Flemming (MDC) and Gina Sorensen for all of their work on scouring the literature in order to document the habitat affinities of each species.

Numerous individuals helped us deal with the difficult issue of trying to account for human stressors and threats. Thanks to Clif Baumer (NRCS), Todd Blanc (MDC), Paul Blanchard (MDC), Charlie DuCharm (MDNR), Victoria Grant (NPS), Valerie Hansen (COE), Robert Harms (MO Natl. Guard), Robb Jacobson (USGS), Pat Kowalewycz (USFS), Charlie Rabeni (MO Coop Unit), Joe Richards (USGS), Mike Roell (MDC), Randy Sarver (MO DNR), Mike Schanta (USFS), Chris Schmitt (USGS), William Schweiger (EPA), Steve Solada (MoDOT), Jodie Staebell (COE), Bob Steiert (EPA), and Rich Wehnes (MDC).

Several individuals from MDC also deserve thanks for their foresight and hard work on the development of the comprehensive freshwater biodiversity conservation plan for Missouri. All of the regional biologists that worked on developing the plan, you know who you are, THANKS! Special thanks to Steve Eder, Dennis Figg, Tim Nigh, Kevin Richards, Lynn Schrader, and Rich Wehnes. Your hard work will not go unnoticed by future generations.

We also need to thank all of the people working on aquatic GAP projects throughout the nation. Your questions and suggestions sharpened our thinking and improved our methods and the final product. We sincerely look forward to our continued collaboration in the future.

Finally, we would like to thank all of the MoRAP partner agencies and their staff that have helped in so many ways with this project, through work on committees, offering assistance or advice, and being a sounding board when difficult decisions had to be made.

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CHAPTER 1

Introduction

In the impending biodiversity crisis, much attention has focused on tropical moist forests, and there is growing interest in ocean conservation. Freshwater systems have received less attention, however, and rivers and streams perhaps least of all. - J. D. Allan and A. S. Flecker

1.1 Background

The National Gap Analysis Program (GAP) was initiated in 1988 to provide a coarse-filter assessment strategy for identifying and prioritizing biodiversity conservation needs (Scott et al. 1993). While GAP has made enormous strides in developing and conducting coarse-filter biodiversity assessments for terrestrial ecosystems, much less has been accomplished for aquatic ecosystems. The program's initial focus on terrestrial vertebrates and vegetation types was a choice based on what was achievable at that early time in the history of the program (Jennings 1999). In principle, GAP is committed to developing biogeographic information and assessment strategies for all major ecosystem types (Jennings 1999).

The need for developing an aquatic component of GAP was recognized as early as 1993, when Congress allocated the funds needed to support such an effort. Those funds, however, were rescinded. GAP still managed to initiate development of an aquatic component of the program in 1995 with the start of a pilot in the upper Allegheny River Basin in Western New York, which was completed in 1999 (Meixler and Bain 1999). In 1997 in cooperation with the Missouri Resource Assessment Partnership (MoRAP) and financial assistance by the USGS National Water Quality Assessment Program, the U.S. Department of Defense and the Missouri Department of Conservation, GAP initiated a statewide pilot project for the state of Missouri. Both of these projects focused on riverine ecosystems. This report summarizes the approach, results and conclusions of the Missouri pilot project.

1.2 How This Report is Organized

This report is a summation of a complex scientific project. Its organization follows the general chronology of the project. It departs from standard scientific reporting by mixing results and discussion within individual chapters. This was done to provide users of the data with a more concise and complete reference for each data and analysis product.

We begin with an overview of freshwater biodiversity in the United States followed by a section, which reviews the GAP mission, concept, and limitations. We then review the principle goal and objectives of this project and the scope/focus of our project. Next is an overview of the information/data requirements for ecologically-based conservation planning in general and more specifically conducting a gap analysis for riverine ecosystems. We then discuss the issue of why we believe it is not advisable to conduct a fully integrated aquatic and terrestrial gap analysis. Next are chapters on the geospatial and tabular datasets that we developed to meet the information/data needs for identifying conservation gaps and subsequently prioritizing opportunities to fills these gaps: a) classifying riverine ecosystems, b) predicting species distributions and biological potential, c) stewardship mapping, and d) accounting for human stressors. Then we provide overview of the methods and results of a series of gap analyses conducted at multiple spatial scales. This series of gap analyses are analogous to those performed in the terrestrial component of gap. Next we cover the methods and results of a statewide freshwater biodiversity conservation plan that was conducted in cooperation with the Missouri Department of Conservation. This chapter illustrates how data generated from an aquatic GAP project can be used in a proactive manner to produce a blueprint for conservation that seeks to fill existing conservation gaps. We then present the methods and results of a more stringent gap analysis for Missouri, which incorporates additional criteria such as representation, connectivity, and present-day environmental quality that are critical for the long-term persistence of freshwater biodiversity. In the last two chapters we discuss, a) how to obtain the data and the appropriate use of the data and b) the training workshops we have held and the publications and presentations we have given pertaining to our work on this project.

1.3 Overview of Freshwater Biodiversity in the United States

Rivers and streams play an important role in shaping and sustaining human existence on earth. They provide critical ecosystem services such as industrial and municipal water supply, renewable energy production, irrigation, flood control, transportation, commercial fisheries, and the assimilation of human wastes (Allan and Flecker 1993; Doppelt et al. 1993). Rivers and streams also have immense recreational value, from “consumptive” uses such as sport fishing, to “non-consumptive” uses such as rafting and canoeing, swimming, streamside hiking, camping and wildlife observation, and the general appreciation of scenic values and aesthetics (Doppelt et al. 1993). The global economic value of these and other services has been estimated to be in the trillions of dollars (Revenga et al. 2000).

At any given time only about 0.01% of the total volume of water on Earth occurs in rivers and lakes. Yet, it has been estimated that anywhere from 25% (Stiassny 1996) to over 50% (Abramovitz 1996) of the global vertebrate diversity is concentrated into this tiny fraction of the biosphere with the vast majority of this diversity occurring within and along riverine ecosystems. Unfortunately, most conservation lands in the United States are situated in the uplands away from these “ribbons” of extraordinary biological diversity due to the fact that the lands adjacent to rivers and streams are the most easily developed and have high economic value for housing, agriculture, or other human uses.

When it comes to freshwater ecosystems the North American continent, and in particular the United States, harbors an astounding proportion of the world’s freshwater species (Warren and Burr 1994; Master et al. 1998; Olson and Dinerstein 1998). Ten percent of all the freshwater fish species, 30% of all the freshwater mussels, and an astounding 61% of all the freshwater crayfish that have been described worldwide are found within the United States (Page and Burr 1991; Williams et al. 1993; Taylor et al. 1996; Master et al.1998). Even more impressive proportions exist for other taxa (e.g., stoneflies, dragonflies, mayflies) (Master et al. 1998). Statistics for these groups are certainly influenced to some degree by global disparities in collection effort afforded these taxa and therefore likely inflate the global distinctiveness of freshwater species richness of the United States. Nonetheless, it is quite apparent, from a global perspective, that the United States is a global "hot spot" for freshwater biodiversity, especially when comparisons are restricted only to temperate regions.

Despite these impressive statistics, North America and the United States are facing a freshwater biodiversity crisis. In just the last one hundred years 123 freshwater animals have gone extinct in North America (Ricciardi and Rasmussen 1999). In the United States alone, 71% of freshwater mussels, 51% of freshwater crayfish and 37% of freshwater fish are currently considered vulnerable to extinction (Williams et al. 1993; Warren and Burr 1994; Taylor et al. 1996; Master et al. 1998). Perhaps even more alarming are the predictions presented by Riccardi and Rasmussen (1999). Using extinction records and an exponential decay model they compared both current and predicted future extinction rates of several taxonomic groups by standardizing these rates according to the size of the species pool. From this analysis they found extinction rates of freshwater fauna in North America to be 5 times higher than those of terrestrial fauna. In addition, by assuming that imperiled freshwater species would not survive throughout the 21st century, their model projects a future extinction rate of 4% per decade, which is comparable to percentages that have been estimated for tropical rain forests.

While much attention has been focused on the global losses of terrestrial biodiversity especially in tropical ecosystems, comparatively little attention has been given to the alarming declines in freshwater biodiversity (Allendorf 1988; Hughes and Noss 1992; Allan and Flecker 1993; Stiassny 1996; Vreugdenhil et al. 2003). A variety of reasons have been given for this lack of scientific and public attention (See Winter and Hughes 1996), however, it is encouraging to see that within the last decade more and more attention has been focused on conserving freshwater biodiversity (Abell et al. 2000, Allan and Flecker 1993; Blockstein 1992; Hughes and Noss, 1992, Stiassny 1996; Ricciardi and Rasmussen 1999). Much of this attention has focused on outlining the severity of the problem, the likely causes for declines, and providing general recommendations for curbing losses of biodiversity in freshwater ecosystems. Yet, as Moyle and Yoshiyama (1994) noted, a critical first step to slowing these losses involves identifying gaps in existing efforts to conserve freshwater biodiversity across the landscape and then prioritizing opportunities to fill these gaps--and this is the overall goal of the USGS National Gap Analysis Program and our project.

1.4 The Gap Analysis Concept

The vast majority of past and present efforts to preserve biodiversity have primarily focused on rescuing individual species, subspecies, or populations from the brink of extinction or local extirpation (Franklin 1993; Scott et al. 1993). This reactive, species-by-species approach to conservation has proved difficult, expensive, biased, and inefficient (Hutto et al. 1987; Scott et al. 1987, 1991; Margules 1989; Noss 1991). Considering the limited human and financial resources dedicated to the recovery of the rapidly growing list of endangered and threatened species it is unlikely that such approaches will slow the rapidly accelerating extinction rates we are currently witnessing (Scott et al. 1993; Wilcove 1993). The existing system of protected areas managed for their natural values represent about 10% of the world's surface area (Vreugdenhil et al. 2003) and only about 3% for the 48 conterminous United States (Scott et al. 1993), which is insufficient to maintain either species diversity or functional ecosystems (Grumbine 1990).

Biological diversity (biodiversity) is the concept around which new concerns about biological conservation are rallied. Biodiversity refers to the variety and variability among living organisms and the environments in which they occur and is recognized at genetic, population, species, community, ecosystem, and landscape levels of organization (U.S. Congress 1987, Noss 1990). The goal of biodiversity conservation is to reverse the processes of biotic impoverishment at each of these levels of organization. Ecological and evolutionary processes ultimately are as much a concern in a biodiversity conservation strategy as are species diversity and composition. Thus, biodiversity conservation represents a significant step beyond endangered species conservation (Noss 1991, Scott et al. 1991). Most significantly, biodiversity conservation is proactive as opposed to reactive last-ditch efforts.

Presuming that a relatively small portion of the total land base will be devoted to biodiversity conservation in the near future, objective techniques are needed to identify and rank proposed conservation areas. Of greatest interest is identification of species, community types, or representative ecosystems not already represented in areas managed exclusively or primarily for the long-term maintenance of populations of native species and natural ecosystem processes. Although a wide variety of conservation evaluation methods have been developed (see Usher 1986), only a few have attempted to assess the conservation value of large geographic areas in a quick and cost-effective manner (e.g., Bolton and Specht 1983, Margules and Austin 1991).

The US Geological Survey’s National Gap Analysis Program (GAP) was initiated in 1988 to provide a coarse-filter approach for identifying biodiversity conservation needs. It seeks to identify gaps in existing conservation efforts that may be filled through establishment of new reserves or changes in land management practices (Scott et al. 1993). Gap Analysis is a technically efficient version of the well-established method of identifying gaps in the representation of biodiversity in biodiversity management areas (Scott et al. 1987, 1989, 1991; Burley 1988; Davis et al. 1990). This approach to conservation evaluation has been widely used in Australia (Specht 1975, Bolton and Specht 1983, Pressey and Nicholls 1991).

1.5 Goals and Objectives

The principal goal of our project was to identify riverine ecosystems and species not adequately represented (i.e., gaps) in the matrix of conservation lands in Missouri. In addition, we wanted to provide spatially explicit data that could be used by natural resource professionals, legislators, and the public to make more informed decisions for prioritizing opportunities to fill these conservation gaps and to devise strategic approaches for developing effective long-term biodiversity conservation plans. Furthermore, as a pilot project for a national program, we also had the goal of developing a broadly applicable gap analysis methodology. We addressed this goal by utilizing nationally standardized and available geospatial data wherever possible and also by devising a flexible conservation assessment methodology, which can accommodate the differences in data availability (e.g., biological) that exists among states across the United States.

The specific objectives of the project were to:

1. Classify and map riverine ecosystems into distinct ecological units at multiple levels.
2. Develop statewide predictive distribution models/maps for all fish, mussel, and crayfish species at the valley-segment scale.
3. Generate local, upstream riparian and overall watershed ownership/stewardship statistics for each valley segment.
4. Account for factors that negatively affect or threaten freshwater biodiversity in Missouri.
5. Assess gaps in the conservation of riverine ecosystems and species at multiple spatial scales.
6. Provide data and information to decision makers that will assist with conservation planning efforts directed toward filling identified conservation gaps.
7. Develop a statewide freshwater biodiversity conservation plan.

1.6 Study Area

Missouri's riverine environments and biota are influenced by both abiotic factors such as climate, geology, landform, and soils, as well as by evolutionary factors resulting from historic and contemporary drainage patterns that have isolated populations and caused faunas to diverge. In the following section we present a broad overview of Missouri with a focus on factors that control the character of streams, followed by more detailed descriptions of the three Aquatic Subregions, the Central Pains, the Ozarks, and the Mississippi Alluvial Basin. These three Subregions are remarkably different in their geologic, topographic, and edaphic features, and these differences are reflected in the relatively distinct freshwater assemblages that exist within each Subregion (Pflieger 1971).

Statewide Overview

Missouri is a physiographically diverse state situated in the east-central United States (Figure 1.1). Two great rivers, the Mississippi, forming the eastern border, and the Missouri, forming the northwestern border and cutting an east-west path across the state to meet the Mississippi, give the state a unique identify. These rivers were both originally formed from the melt waters of continental ice sheets. The Missouri River roughly forms the southern boundary of Pleistocene continental ice sheets in the state, and streams to the north often originate in glacial tills or loess.

Figure 1.1. Map of Missouri showing the principal drainage systems and the three Aquatic Subregions that account for major differences in instream habitat and freshwater assemblages across the state.

The Aquatic Subregions of Missouri are separated along drainage divides that generally correspond with abrupt transitions in geology, landform, soils, landcover, and groundwater influences (Figures 1.2-1.6). The glaciated plains north of the Missouri River together with the unglaciated Osage Plains to the southwest form the Central Plains Aquatic Subregion. In southeastern Missouri, the Mississippi embayment forms the Mississippi Alluvial Basin Subregion. Finally, the Ozark Aquatic Subregion, which lies between these two Aquatic Subregions, is a dissected plateau underlain mainly by Mississippian, Ordovician, and Cambrian dolomites and limestones.

In a broad context, the climate of Missouri is continental and strongly seasonal. Both precipitation and mean temperatures primarily vary along a gradient extending from the northwest to the southeast. The average annual temperatures range from over 58º F in the southeast to under 52º F in the northwest. Mean annual precipitation ranges from more than 50 inches in the southeast to less than 36 inches in the northwest. Pre-European upland vegetation was generally prairie or savanna on flat uplands and woodland or forest in the more rugged landscapes. Forests were most extensive in the relatively wet and rugged southeastern Ozarks. High summer temperatures combined with periodic drought lead to substantially low base flows and relatively high temperatures in many streams throughout the state. These physiologically stressful periods have helped shape the native riverine biota, especially in the Central Plains (Pflieger 1971).

Figure 1.2. Map showing system-level geologic differences among the three Aquatic Subregions of Missouri.

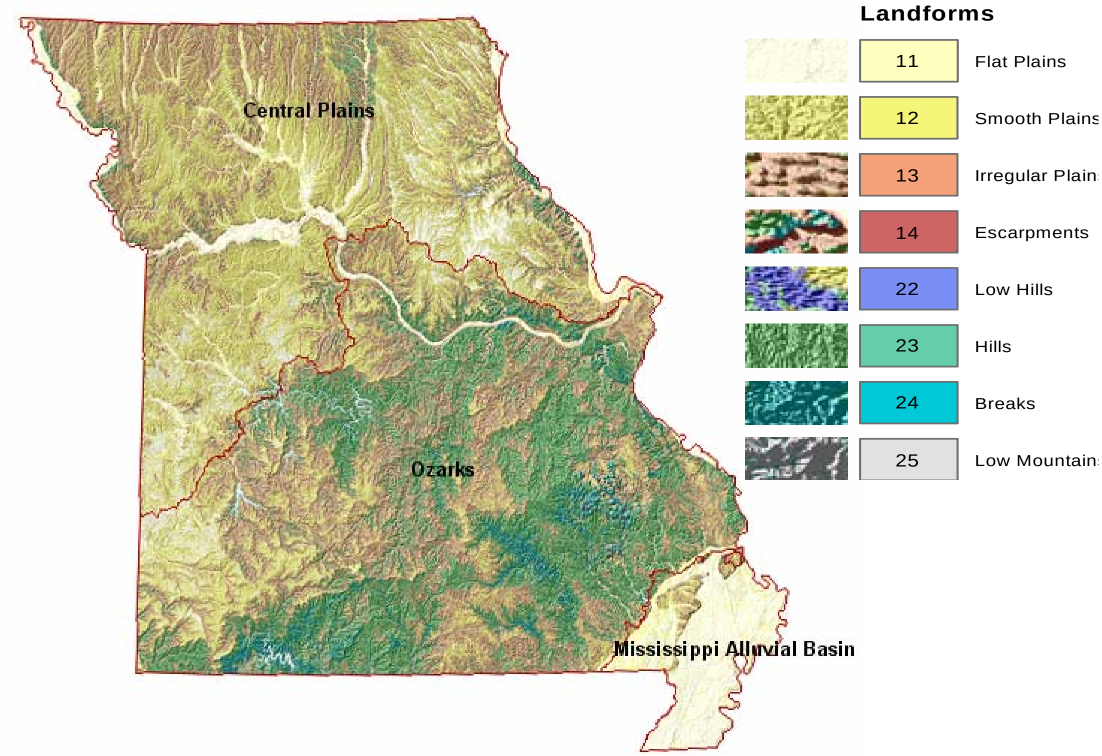


Figure 1.3. Landforms of Missouri, generated from a 30-meter Digital Elevation Model, illustrating the major differences in topography and relief among the three Aquatic Subregions in the state.

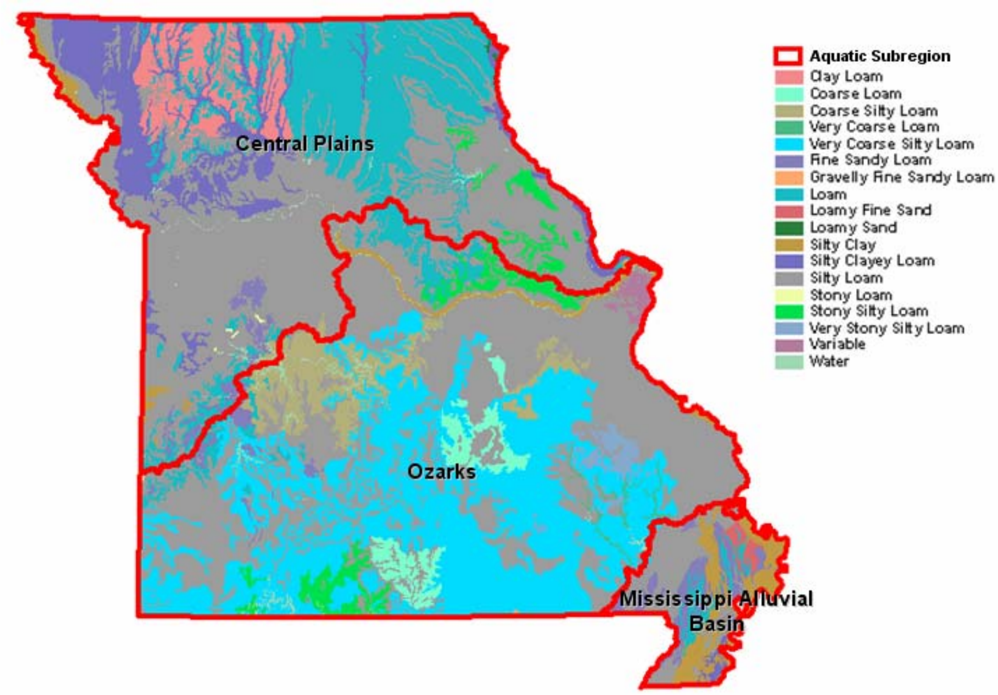


Figure 1.4. Map of soil surface textures across Missouri, based on STATSGO soils coverage.

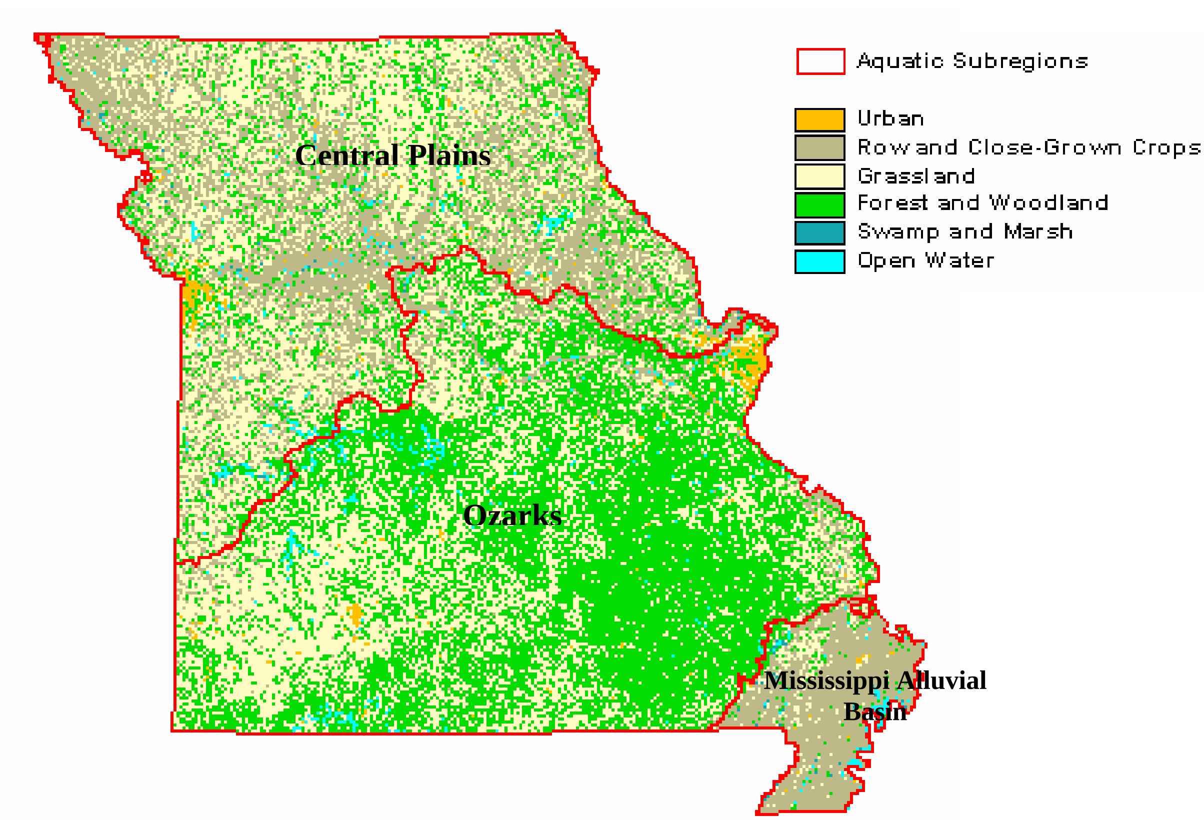


Figure 1.5. Landcover map (circa 1992-93) of Missouri illustrating differences among the three Aquatic Subregions.

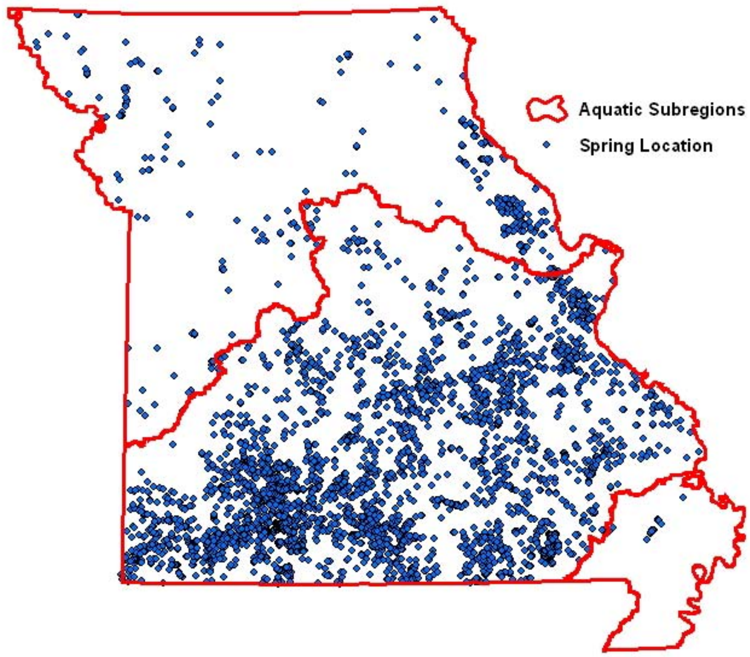


Figure 1.6. Map showing the distribution of springs in Missouri and the pronounced differences in the presence and density of springs among the three Aquatic Subregions.

The conversion of grassland to row crops, which now comprise 25% of the state, has increased sediment and nutrient loads across much of Missouri, especially north of the Missouri River (see Figure 1.5). Likewise, urban land now comprises almost 5% of the landscape, and has led to significant changes in the biophysical character of streams in areas of major metropolitan development, including Kansas City, St. Louis, and Springfield. Large impoundments have been constructed on several major streams throughout Missouri; especially in the central and southwestern Ozarks. It has also been estimated that approximately 300,000+ small artificial waterbodies (less than 2.5 acres) have been constructed in Missouri (Vandike 1995; Smith et al. 2002). The vast majority of these occur in the Central Plains Aquatic Subregion (Pflieger 1971; Nigh and Schroeder 2002). The ecological effects of these artificial waterbodies include the expansion of predatory game species (e.g., largemouth bass and bluegill) into entire regions or watersheds and more locally into headwater streams where they historically did not occur (Pflieger 1997), increased evaporation rates, diversion and delay of the downstream transmission of water, and altered biochemical reactions and groundwater interactions (Smith et al. 2002).

Central Plains

Glacial loess and till, generally thinning to the east and away from the big rivers, covers most of the Central Plains north of the Missouri River. Shale, limestone, and sandstone of the Osage Plains underlie the Central Plains south of the Missouri River, and low, northeast to southwestern trending scarps have formed where limestone and sandstone are exposed (see Figures 1.2). North of the Missouri River, elevation tends to increase to the north. The Grand Divide in east central Missouri north of the Missouri River, which reaches elevations of 1000 feet, separates streams that flow eastward to the Mississippi from those that flow into the Missouri (see Figure 1.1). The latter generally flow southward and form parallel drainage patterns on the landscape in contrast to dendritic drainage patterns of watersheds to the south of the Missouri River. Deep loess deposits blanket the northeastern fifth of Missouri, and give unique character to landscapes and watersheds such as the Platte and Nodaway Rivers. Loess hills are also found along the Missouri River and to a lesser extent along the Mississippi (see Figure 1.4). Loess deposits thin to the east, and the most extensive nearly flat landscape in Missouri, the Claypan Till Plains, extends from the Grand Divide eastward to the Mississippi River hills. To the west of this divide in north central Missouri, the Chariton River Hills represent the most extensive hilly landscape in the Central Plains.

Landscapes of the Central Plains are mainly flat to gently sloping with an average land slope of 5% and local relief from 20 to 200 feet (see Figure 1.3). Average stream gradients are 10.3 m/km for headwaters, 2.3 m/km for creeks, 0.7 m/km for small rivers, and 0.3 m/km for large rivers (Figure 1.7). Streams tend to occupy broad valleys and grade gradually into uplands, especially in the southwest and in the east central portions of the subregion. Substrates are generally fine silts and sands and streams are frequently turbid. Historically, headwaters often had well defined pools and riffles and downstream pools were long and riffles were short or absent. Few large springs exist and hence base flows can be low and smaller streams are often intermittent (see Figure 1.6). Low dissolved oxygen concentrations are common throughout the region, and helped shape the biota. Many streams have been straightened and channelized in modern times, grasslands have been converted to fertilized cropland, and streamside gallery forests have been removed (see Figure 1.5). Hence today's streams probably are more turbid, tend to have lower dissolved oxygen concentrations, have less predictable base flows, and have wider temperature fluctuations that they did in pre-European settlement times.

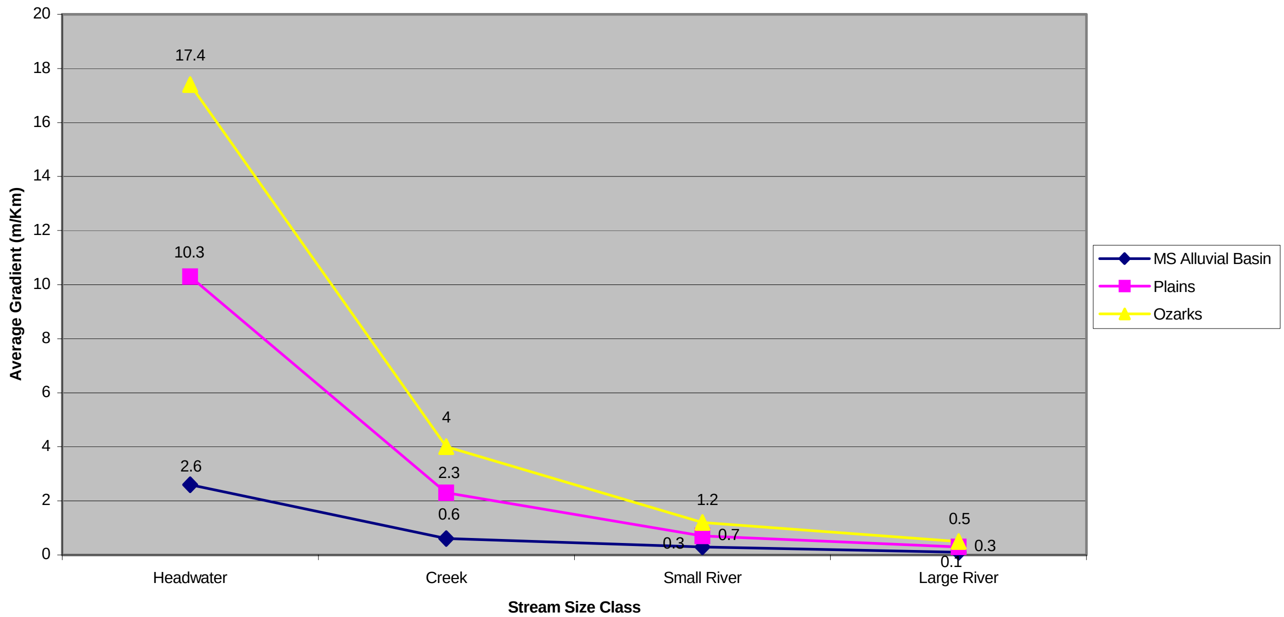


Figure 1.7. Comparison of average stream gradients (m/km) for four stream size classes within each of Missouri's Aquatic Subregions.

Ozarks

Higher ground is situated generally southwest to northeast across the Ozarks, with the highest point at Tom Sauk Mountain, at 1772 feet, in the southeastern Ozarks. The central divide creates north-slope streams that flow mainly to the Missouri River and south-slope streams that flow toward the Mississippi River (see Figure 1.1). The exceptions to this pattern are in the southwestern Ozarks, where the Spring, Elk, and Neosho Rivers flow west toward the Grand (Arkansas) River system. Hence, streams that share a common drainage divide may be separated by hundreds or even thousands of stream miles, including portions of the Missouri and Mississippi Rivers, which are effective dispersal barriers for much of the upstream biota. Relatively high, flat plateaus separate hills and breaks landscapes carved by major streams (see Figure 1.3). Most of the plateaus are dolomite or limestone, but some watersheds, such as the James and Bourbeuse Rivers, have higher percentages of sandstone which tend to make their riverine habitats distinct within the Ozarks (see Figure 1.2). In the southeastern Ozarks, Precambrian granite is exposed and gives unique character to steams, especially in the St. Francois River watershed. In general, the streams of the Ozarks are less highly impacted by gross physical alterations such as agriculture and channelization versus the other Aquatic Subregions in Missouri, and land cover is mainly in a semi-natural state (<10% urban plus cropland) (see Figure 1.5). Ironically, some of the poorest water quality in the state can be found in the Ozarks in areas downstream from urban development in the St. Louis area, and stream segments downstream from lead mines sometimes have extensive areas with substrates derived from mine tailings (Cieslewicz 2004).

The Ozarks are more rugged than the Central Plains; with average land slope of 9% and local relief over 300 feet common (see Figure 1.3). Smaller streams tend to be relatively high gradient, with an average of 17.3 m/km for headwaters and 4 m/km for creeks (see Figure 1.7). Headwaters are characterized by well-defined riffles and short pools over gravel, cobble, or bedrock substrates, whereas creeks have riffles over gravel or cobble and pools with sand and silt overlying larger substrates. Small river gradients average 1.2 m/km and larger rivers average 0.5 m/km, only slightly higher than for other Aquatic Subregions. Small rivers are characterized by deep pools with silty substrates and riffles composed of gravel and cobble. Deep pools have developed along limestone bluffs rising vertically from some small rivers, forming a unique and beautiful habitat feature. Large rivers are characterized by long pools and deep chutes along with backwaters and cut-offs. Pools have sand and silt bottoms, while swifter areas maintain gravel and cobble substrates, except for those streams directly entering the Missouri of Mississippi Rivers (e.g. Meramec and Gasconade). The relatively rugged landscape accounts for high peak discharges in Ozark streams. The permeable dolomite and limestone bedrock allow for the formation of karst that supports numerous springs in many parts of the Subregion, and these springs account for relatively stable base flows and wide temperature variation among stream reaches (see Figure 1.6). Selected Ozark streams systems, including the Current, Meramec, and Gasconade watersheds, remain free flowing (without major reservoirs) and occur in a wooded, relatively rugged, semi-natural landscapes, making them among the most aesthetically appealing and ecologically intact stream ecosystems in the state (see Figure 1.5).

Mississippi Alluvial Basin

The Mississippi Alluvial Basin (MAB) in Missouri represents the northern extension of the broad valley of the Mississippi River. The natural character of the streams and vegetation, including slow-moving, meandering streams on a great river floodplain with wet prairies, marshes, swamps, and bottomland hardwood forests, has been entirely altered by land clearing and the installation of an amazingly complex network of drainage ditches (see Figure 1.1).

The MAB is a nearly flat plain with natural levees and meander scars except for Crowley's ridge, which is a narrow band of hills formed as an erosional remnant (see Figure 1.3). Bedrock is an unimportant feature of MAB landscape except within Crowley’s Ridge, which is underlain mainly by Cretaceous and Tertiary sandstones, siltstones and shales with some minor inclusions of Ordovician sandstones and dolomites (see Figure 1.2). Crowley’s Ridge is capped by a relatively thick mantle of windblown loess deposits similar to those found along the bluffs of the Missouri and Mississippi Rivers in other parts of the state (Pflieger 1971). The remainder of the MAB is underlain by Cretaceous and Tertiary deposits of clay, sand, and gravel that range from a few feet to more than 2,700 feet in thickness (Grohskopf 1955). These older sediments are buried under a layer of alluvium deposited by the St. Francis, Mississippi, and Ohio rivers during Pleistocene and recent times (Pflieger 1971) (see Figures 1.2 and 1.4).

In its original condition the MAB was one of the most heavily timbered regions of Missouri (Pflieger 1971). Also, despite the nearly level landscape of this Subregion, a relatively high water table combined with varied soils provided a diverse landscape for plant communities to form. Site conditions within the MAB ranged from permanently flooded areas supporting only emergent or floating aquatic vegetation, to high elevation sites supporting complex hardwood forests (Brown et al. 1999). Of all the regions of Missouri, the MAB has lost the greatest part of its historic vegetation with only a few small remnants of the nineteenth century forest cover remaining (Nigh and Schroeder 2002). Almost 95% (excluding Crowley's Ridge) of this Subregion has been drained and converted to farmland with the vast majority being cropland; particularly soybeans, wheat, corn, cotton, and rice (see Figure 1.5).

Average annual runoff ranges from 18 to 20 inches, which is the highest in the state. However, the nearly flat topography of the MAB results in low runoff rates and the sand and gravel alluvial deposits that overlay the relatively impermeable clayey subsoils make excellent shallow aquifers (Pflieger 1971). These two factors are collectively responsible for the relatively stable hydrographs and high baseflow potential of streams and ditches within the MAB where even the smallest channels tend to carry water during the driest periods of the year. However, springs are relatively scarce except along the toeslope of Crowley's Ridge (see Figure 1.6).

The ditches and few remaining natural streams in the MAB vary substantially in terms of discharge, turbidity, current, substrates, aquatic vegetation and shading by riparian vegetation (Pflieger 1971). Smaller ditches are most variable in character, but generally have higher water clarity than larger ditches. Some have no perceptible current during base flow with bottoms comprised mainly of silt while others are fairly swift and have bottoms mostly comprised of sand and small gravel (Pflieger 1989). Channels with clear water and little riparian shading are generally choked with submergent vegetation. Some of the major ditches are large enough to be classified as either small or large rivers. These ditches are extremely wide and shallow with considerable current throughout. Channel gradients are significantly lower in the MAB than the other two Subregions (see Figure 1.7). Channels classified as headwaters have an overall average gradient of 2.6 m/Km, while the average gradient of channels falling within all other sizes classes are less, and often substantially less, than 1 m/Km. Despite these low stream gradients headcutting and rill and gully erosion are substantial problems upstream from channelized sections (Boone 2001). Cover is generally sparse and is often confined to undercut banks and associated vegetation or woody debris. Woody cover is typically much more abundant in unchannelized stream sections (Boone 2001).

1.7 Focus of the Missouri Aquatic GAP Project

The goals and objectives of our project are by no means small objectives. Consequently, we had to establish some priorities to make the project more reasonable in scope and to help maintain a more structured approach to our efforts. First, as evidenced by information in the preceding sections and our project objectives, we strictly focused on riverine environments, exclusive of the Missouri and Mississippi Rivers. Missouri is essentially a "stream state" and most of our aquatic biodiversity concerns are centered in riverine ecosystems (Pflieger 1989). Second, although it is envisioned that the aquatic component of GAP will ultimately entail holistic assessments for all major aquatic taxa, our project focused primarily on *fish, mussels, and crayfish*. Explicitly focusing on these three taxa was a result of the availability and quality of existing collection data.

1.8 Why We Believe the Aquatic and Terrestrial Components Cannot be Integrated a Priori

The title of this section pertains to the most commonly asked question posed to those of us working on aquatic GAP projects. This is certainly an important question, because ideally we would like to believe that all elements of biodiversity could ultimately be integrated into a single assessment of conservation gaps and opportunities. We admit that we had these same aspirations when we began our project and held this belief for a very long time. However, we began to realize that even though the basic goal and objectives of the terrestrial and aquatic components of gap are indeed the same, there is a major obstacle to such upfront integration.

The foremost obstacle to a fully integrated terrestrial and aquatic gap analysis pertains to the fact that if we are going to conserve biodiversity we must conserve ecosystems (Franklin 1993; Grumbine 1994). Traditionally, ecoregions have served as the geographic framework for defining terrestrial ecosystems and conserving terrestrial biodiversity. While ecoregions do a good job of accounting for structural and functional differences in freshwater ecosystems, they do not account for important compositional differences (species and genetic composition) resulting from the isolation of freshwater faunas largely related to historical and contemporary drainage patterns (Figure 6) (Pflieger 1971; Matthews 1998). Also, in most instances, ecoregions do not define interacting systems, which is a fundamental concept found in virtually every definition of an ecosystem. Watersheds or drainages, on the other hand, do define interacting systems and do act as a principle evolutionary and distributional constraint for freshwater organisms. Major drainage systems are analogous to islands embedded within the landscape. Our approach to freshwater biodiversity conservation must therefore be similar to the approach taken to conserve biodiversity on a chain of islands, where each island must be treated as a distinct entity.

Consequently, defining ecosystems in freshwater environments requires the integration of ecoregion and drainage boundaries. Ironically, in most instances watershed boundaries play only a marginal role in the defining interactive systems for terrestrial environments, except in mountainous regions. This dichotomy is a critical fundamental difference that dictates the use of different geographic frameworks for conserving freshwater and terrestrial biodiversity. This is why we developed a separate aquatic ecological classification framework for our project. This fundamental difference should not be viewed as an impediment to conserving biodiversity. We like to say that we have "geography on our side." Conservation assessments or gap analyses can be performed separately for terrestrial and aquatic ecosystems and the results then be spatially integrated a posteriori into an overall assessment or analysis.

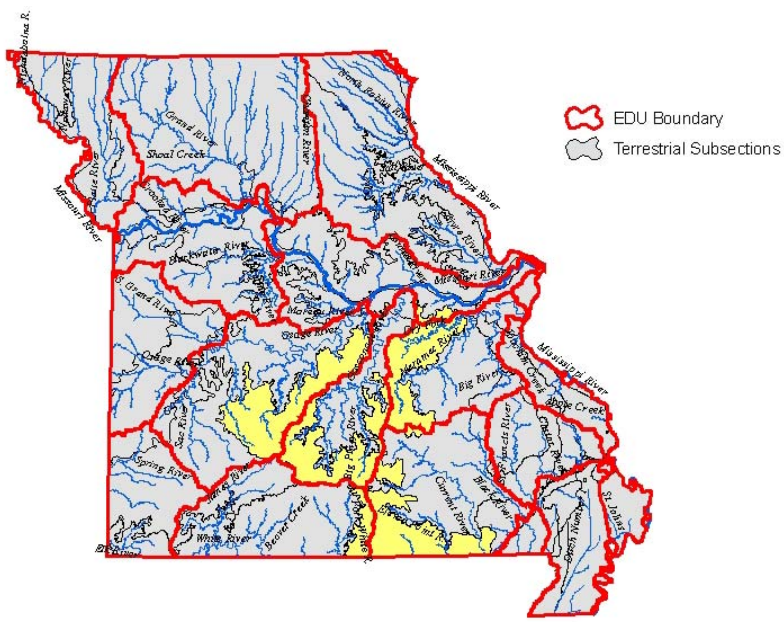


Figure 1.8. Map showing how terrestrial ecoregions do not account for important evolutionary constraints that partially determine the composition of freshwater assemblages. The Ozark/Central Plateau Ecological Subsection (Nigh and Schroeder 2002; in yellow) crosses five major drainages (EDUs) within Missouri. Even though the physicochemical character of the streams across the Ozark/Central Plateau are relatively similar, the local assemblages that inhabit the streams within this ecological subsection differ across the major drainages due to the different evolutionary histories of these drainages.

CHAPTER 2

An Overview of Biodiversity Conservation Planning

Before beginning, plan carefully. - Marcus T. Cicero

2.1 Purpose of Overview

Before discussing the specific data we compiled or developed for the Missouri Aquatic GAP Project, we believe it necessary to provide an overview of conservation planning. This overview provides the context needed to more clearly illustrate why we developed each geospatial datalayer. Margules and Pressey (2000), Groves (2003), and Noss (2004) all provide excellent overviews of conservation planning and we essentially cover the most basic elements discussed by these authors in our review of the topic.

2.2. Establishing Goal and Fundamental Principles, Theories, and Assumptions

The first step in conservation planning is to establish a goal expressing the focus of the effort. This should not be confused with the quantitative conservation goals that are established when devising a specific conservation strategy (see below). Goals pertaining to biodiversity conservation have been variously described, but most reflect the need to conserve and restore the *processes* that generate or sustain biodiversity.

Once a goal has been established, the fundamental principles, theories, and assumptions that must be considered in order to achieve this goal must be identified. These generally pertain to basic ecological or conservation principles and theories that provide the foundation of the overall conservation strategy that will be used for achieving the overall goal. Examples include:

- In order to conserve biodiversity we must conserve ecosystems. Or, in order to conserve or restore the biological assemblage of a particular area of interest we must take measures to conserve or restore the critical structural features, and functional and evolutionary processes that support this assemblage (Franklin 1993; Grumbine 1994; Leslie et al. 1996; DeLeo and Levin 1997).
- Biodiversity can be described and should be conserved at multiple levels of organization (Whittaker 1962, 1972; Franklin 1993; Noss 1994; Jennings 1996; Leslie et al. 1996).
- Populations, not species, are the fundamental unit of conservation (Leslie et al. 1996; Meffe and Carroll 1997).
- Biodiversity conservation efforts should focus on identifying and collectively conserving the variety of distinct genotypes, populations, species, communities,

assemblages, and ecosystem types across the landscape (Angermeier and Schlosser 1995; Grossman et al. 1998; Olson and Dinerstein 1998; Abell et al. 2000).

- Proactive protective measures are less costly and more likely to succeed than restoration actions (Scott et al. 1993).
- Protected areas are critical to the long-term conservation of biodiversity (Rodrigues et al. 2003).
- We cannot directly measure, map, or conserve biodiversity, but we can measure, map, and conserve surrogate biotic and abiotic conservation targets (Margules and Pressey 2000; Roux et al. 2002; Noss 2004).
- Taking measures to conserve a variety of biotic and abiotic targets is the best and most efficient approach to conservation (Kirpatrick and Brown 1994; Noss et al. 2002; Diamond et al. 2005).
- The structural features and functional processes of a particular location, and how they change through time, provide the habitat template upon which ecological strategies of species develop and evolve through time (Southwood 1977).
- Connectivity among habitats is often essential for meeting the various life history requirements of certain species, as well as, providing essential dispersal avenues during periods of disturbance (Schlosser 1987; Schlosser 1995; Matthews 1998; Fausch et al. 2002; Rabeni and Sowa 2002; Benda et al. 2004).
- Redundancy in representation of populations or ecosystem types is a safeguard against extinction and also promotes the generation of biodiversity through processes like adaptive radiation, random genetic mutations, and genetic drift (Noss and Cooperrider 1994; Meffe and Carroll 1997; Shaffer and Stein 2000; Groves 2003).
- Priorities should be established and conservation actions taken at multiple spatial scales because different species perceive or utilize the landscape (riverscape) differently and because the critical structural features and functional processes change with the scale of interest (Frissell et al. 1986, Wiens 1989; Angermeier and Schlosser 1995).
- Public ownership does not equate to effective biodiversity conservation, especially in riverine ecosystems (Benke 1990; Allan and Flecker 1993).
- Due to the inherent complexity and dynamic nature of ecosystems, uncertainty is a fundamental component of ecosystem management. This is not an excuse for inaction, but efforts to document and overcome this uncertainty must be a priority (Leslie et al. 1996).
- Because of competing societal demands and the limited human and financial resources dedicated to biodiversity conservation we must recognize that we cannot conserve everything, in fact, in many instances we can only conserve a relatively small fraction of the resource base (Scott et al. 1993; Rodrigues et al. 2003).
- We must therefore strive for efficiency in our conservation efforts and one way to accomplish this is to prioritize locations for conservation and try and maximize the complementarities of protected or focus areas (Margules and Pressey 2000).

This list is long, however, it is by no means complete, and the point here is to show the shear number and complexity of things that must be considered in the conservation planning process. By extension, these same principles, theories, and assumptions should also be considered when trying to identify and develop the data/information that will be most useful to the conservation planning process.

2.3 Selecting a Suitable Geographic Framework

Because conservation planning is a geographical exercise, the next step in the process involves selecting a suitable geographic framework. More specifically, this involves selecting, defining, and mapping *planning regions* and *assessment units*. A planning region refers to the area for which the conservation plan will be developed. It defines the spatial extent of the planning effort(s). Assessment units are geographic subunits of the planning region. These units define the spatial grain of analysis and represent those units among which relative quantitative or qualitative comparisons will be made in order to select specific geographic locations as priorities for conservation. Planning regions and assessment units can be variously defined and should be hierarchical in nature to allow for multiscale assessment and planning (Wiens 1989). Boundaries could be based on sociopolitical boundaries (e.g., nations, states, counties, townships), regular grids (e.g., UTM zones or EPA EMAP hexagons), or ecologically defined units (e.g., watersheds or ecoregions). Since biodiversity does not follow sociopolitical boundaries or regular grids, whenever possible, planning regions and assessment units should be based on ecologically defined boundaries since these boundaries provide a more informative ecological context (Bailey 1995; Omernik 1995; Leslie et al. 1996; Higgins 2003).

2.4 Selecting Surrogate Conservation Targets

Because it is impossible to directly measure or map biodiversity, surrogate targets for conservation must be identified and mapped (Margules and Pressey 2000; Noss 2004). For the terrestrial component of GAP these surrogates generally include plant communities or vegetation types and vertebrate species (Scott et al. 1991). The assumption here is that by taking measures to conserve these surrogates we are in fact taking measures to also conserve those unmapped or unmappable elements of biodiversity. Because different targets often lead to different answers on which locations should be a priority for conservation, it is generally more effective to use a variety of targets (Kirpatrick and Brown 1994; Noss 2004; Diamond et al. 2005). Also, because biological survey data are often incomplete, biased, or completely lacking, abiotic targets (e.g., ecosystems, landscapes, or habitats), which are usually easier to map, are often considered as targets (Belbin 1993; Nicholls et al. 1998; Noss et al. 2002; Noss 2004). Angermeier and Schlosser (1995) and Noss (2004) provide excellent discussions on the reasons for using both biotic and abiotic surrogates. Also, a study by Kirpatrick and Brown (1994) revealed that using both biotic and abiotic targets would likely be the most successful approach to representing the range of biodiversity within a planning region.

2.5 Devising an Overall Conservation Strategy

Once planning regions, assessment units, and conservation targets have been identified and mapped, an overall conservation strategy for selecting priority areas within the planning region must be established. This strategy is built around the fundamental principles, theories, and assumptions that deal with issues such as: How many occurrences of each target should be captured? How much area or length should be captured? Is connectivity essential? If given a choice, should you select locations within existing public lands? Are you interested in selecting relatively high-quality locations for proactive protection efforts or the worst-case scenario for restoration efforts? Unfortunately, for most of these and other pertinent questions there are no detailed guidelines, and even when there is some guidance (e.g., biogeography theory, population viability analysis, or metapopulation theory) the data needed for these more detailed evaluations are usually lacking (Margules and Pressey 2000; Groves 2003). Expert opinion will therefore often play a major role in developing the overall conservation strategy.

In addition to establishing a general conservation strategy, quantitative and/or qualitative assessment criteria, that will be used to make relative comparisons among assessment units, must also be established. These criteria include measures of relative significance or irreplaceability, condition, future threats, costs, and opportunities, which guide the selection of one particular assessment unit over another (Groves 2003). These criteria should also be based upon the previously established fundamental principles, theories, and assumptions.

<u>Examples include</u>

Significance:	species richness, number or percent of endemic species, diversity of habitats, presence of unique habitats, species, communities, or processes
Condition:	percent urban or agriculture, road density, degree of fragmentation, extent of channelization, degree of hydrologic modification, mine density, etc.
Future threat:	recent or projected population trends, potential for future extractive uses
Costs:	acquisition cost, restoration cost, loss of socioeconomic benefits
Opportunities:	leveraging of funds or cooperation among stakeholders, local interest or involvement, ability to receive federal, state, or local funding

After addressing the issues discussed above, the next step involves selecting priority locations within the planning region(s).

Since conservation planning is a geographical exercise, it is no surprise that Geographical Information Systems (GIS) are an invaluable tool. However, because not all of the essential data are in a geospatial format, and because much of the data that are available often lack the necessary detail, expert knowledge must often be incorporated into the planning process. The GIS data provide a more objective, spatially explicit, and comprehensive view of the planning region, while the experts may provide additional and more detailed information for certain locations.

Conservation planning is also a logistical exercise, and once priority areas have been identified, much work remains to be done. The questions of Who? What? How? When? and Why? must all be addressed. Questions such as: Who owns the land within and around each priority area? Who is responsible for implementing on-the-ground conservation actions? What are the critical structural features, functional processes, and species or communities of concern within each priority area? What are the principal threats that must be addressed within each priority area? What are the principal uncertainties surrounding the selection of each priority area and the associated threats and management options? How are we going to eliminate or minimize threats? When should conservation actions be taken, immediately or is there time? Why was each priority area selected, and why is one more "important" than another? Addressing these questions is often more difficult than building the geospatial data sets and associated tools used to select priority areas. However, not addressing these important questions could lead to failure in our efforts to conserve biodiversity (Margules and Pressey 2000). Once these logistical questions have been addressed, then on-the-ground conservation actions can be taken. Monitoring programs must also be established to ensure that conservation efforts are successful and to signal when and possibly how management actions should be modified. Because of the complexity and dynamic nature of ecosystems, adaptive management will be a key to long-term conservation of biodiversity (Leslie et al. 1996).

2.6 Discussion

So, what does this abbreviated overview of conservation planning have to do with the Missouri Aquatic GAP Project? Well, in order to adequately assess gaps in biodiversity conservation we must first identify what constitutes a gap and the only way to do this is to develop criteria for what constitutes "effective" conservation. These very criteria are established in the conservation planning process. Building on the solid foundation of the terrestrial component of GAP and going through the above process were the two most influential factors that guided the decisions we faced about the data to be compiled or developed as well as the overall approach to the Missouri Aquatic GAP Project.

We certainly hope that this overview will also benefit other gap analysis projects. We encourage all GAP practitioners to critically examine each element of conservation planning and use it to help guide decisions surrounding the geospatial data that is developed during each GAP project and the resulting gap analysis. As you read the remainder of this report, we encourage you continually refer back to this chapter; the context it provides helps simplify this very complex project.

Chapter 3

Hierarchical Classification of Riverine Ecosystems

Good classifications make discoveries possible, and, in turn, discoveries change our ways of classifying the things we study. – M. Goldstein and I. F. Goldstein

3.1 Purpose

- Provide the ecological and evolutionary context necessary for making truly relative comparisons among two or more locations.
- Provide an ecologically meaningful geographic framework for conservation planning (i.e., planning regions and assessment units).
- Provide surrogate abiotic conservation targets to complement biotic targets.
- Account for broader ecosystem or evolutionary processes that are often not considered with the use of species data alone.
- Account for poorly known or unknown ecosystem processes, aquatic assemblages, and organisms.
- Provide a geographic template and predictor variables for developing predictive species distribution models and maps.
- Provide the necessary reductionist tool for generating inventory statistics, conducting conservation assessments, and developing conservation plans.
- Enhance our understanding of the number and spatial distribution of distinct ecosystem types and riverine assemblages.
- Enhance communication among resource professionals, legislators, and the public.

3.2. Introduction

It is widely accepted that to conserve biodiversity we must conserve ecosystems (Franklin 1993; Grumbine 1994). It is also widely accepted that ecosystems can be defined at multiple spatial scales (Noss 1990; Orians 1993). Following this logic, a key objective of our project was to define and map distinct riverine ecosystems (often termed ecological units) at multiple levels. However, before distinct riverine ecosystems could be classified and mapped, the question "What factors make an ecosystem distinct?" had to be answered.

Ecosystems can be distinct with regard to their *structure, function, or composition* (Noss 1990). Structural features in riverine ecosystems include factors such as depth, velocity, substrate, or the presence and relative abundance of habitat types. Functional properties include flow regime, thermal regime, sediment budgets, energy sources, and energy budgets. Composition can refer to either abiotic (e.g., habitat types) or biotic factors (e.g., species). While both are important, our focus here will be on biological composition, which can be further subdivided into *ecological composition* (e.g., physiological tolerances, reproductive strategies, foraging strategies, etc...) or *taxonomic composition* (e.g., distinct species or phylogenies) (Angermeier and Schlosser 1995). Geographic variation in ecological composition is generally closely associated with geographic variation in ecosystem structure and function. For instance, fish species found in streams draining the Central Plains of northern Missouri generally have higher physiological tolerances for low dissolved oxygen and high temperatures than species restricted to the Ozarks, which corresponds with the prevalence of such conditions within the Central Plains (Pflieger 1971; Matthews 1987; Smale and Rabeni 1995a, 1995b). Differences in taxonomic composition, not related to differences in ecological composition, are typically the result of differences in evolutionary history between locations (Mayr 1963). For instance, differences among biological assemblages found on islands despite the physiographic similarity of the islands.

Considering the above, our more specific objective was to identify and map riverine ecosystems that are relatively distinct with regard to ecosystem structure, function, and evolutionary history at multiple levels. To accomplish this, an eight-level classification hierarchy was developed in cooperation with personnel from The Nature Conservancy's Freshwater Initiative (Higgins 2003; Higgins et al. 2005) (Figure 3.1). Levels within the hierarchy were either empirically delineated using biological data or delineated in a top-down fashion using landscape and stream features (e.g., drainage boundaries, geology, soils, landform, stream size, gradient, etc.) that have consistently been shown to be associated with or ultimately control structural, functional, and compositional variation in riverine ecosystems (Hynes 1975; Dunne and Leopold 1978; Matthews 1998). More specifically, levels 1-3 and 5 account for geographic variation in *taxonomic or genetic-level composition* resulting from distinct evolutionary histories, while levels 4 and 6-8 account for geographic variation in ecosystem structure, function, and *ecological composition* of riverine assemblages (Table 3.1). The most succinct way to think about the hierarchy is that it represents a merger between the different approaches taken by biogeographers and physical scientists for tesselating the landscape into distinct geographic units.

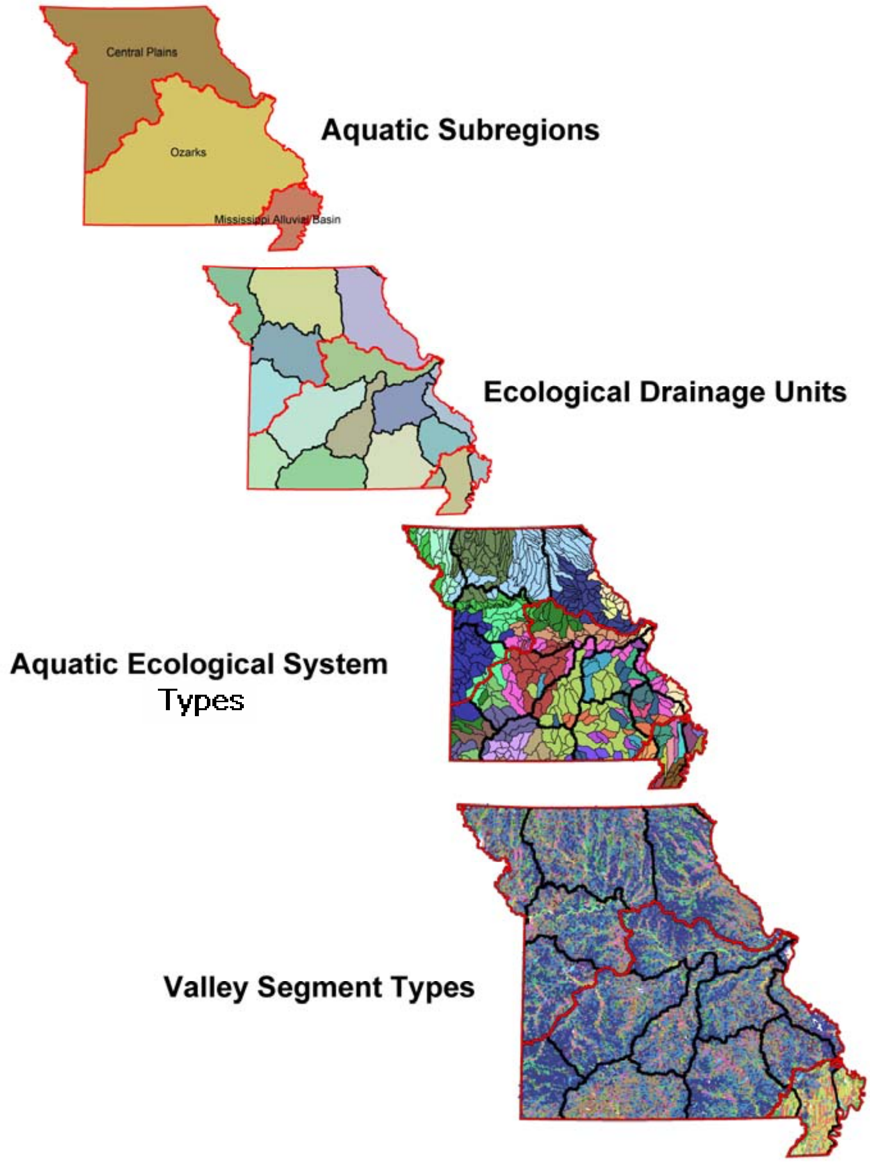


Figure 3.1. Maps showing Levels 4-7 of the MoRAP Aquatic Ecological Classification hierarchy.

Table 3.1. Hierarchical framework, with defining physical and biological features, used for classifying and mapping riverine ecosystems in the Missouri Aquatic GAP Project. Hierarchy is adapted after the classification hierarchies of Frissell et al. 1986, Pflieger et al. 1989, and primarily Maxwell et al. 1995, Seelbach et al. 1997 and Higgins et al. 2005. *Note: Levels in red primarily account for differences in local stream assemblages resulting from distinct evolutionary histories, while levels in black account for differences resulting from geographic variation in ecosystem structure and function.*

Level	Description	Defining Physical Features	Defining Biological Features
Zones	Six major zoogeographic zones of the world that resulted from distinct evolutionary histories associated with plate tectonics	Continental boundaries; Global climate	Family level patterns; Endemism
Subzones	Subcontinental zoogeographic strata with relatively unique aquatic assemblages created in large part by plate tectonics, glaciation, and mountain building	Major river networks and basin boundaries; Regional climate	Family level patterns; Endemism
Regions	Subzone zoogeographic strata created in large part by drainage network patterns that determine dispersal routes and isolation mechanisms that have resulted in different responses to longterm changes in climate	Major river networks and basin boundaries; Regional climate	Family and species level patterns; Endemism; Phylogenetics
Subregions	Region stratification units. Large areas of similar climate and physiography that often correspond to broad scale patterns in dominant vegetation	Regional climate; Physiography; General physiognomy of vegetation	Family and species level patterns; Endemism; Distinct foraging, reproductive and habitat-use guilds; Distinct physiological tolerances and ecomorphologies
Ecological Drainage Units	Subregion zoogeographic strata. Aggregates of drainages within a distinct physiographic setting that share a common evolutionary history	Drainage boundaries; Physiography	Family and species level patterns; Endemism; Phylogenetics
Aquatic Ecological System Types	Hydrogeomorphic subunits of Ecological Drainage Units. Hydrologic units with similar physiographic character, basin morphometry and position within the larger drainage (e.g., located in the headwaters versus near the drainage outlet).	Watershed boundaries; Position within larger drainage; Local and watershed physiography; Local climate (in montane regions); Basin morphometry	Species level patterns; Distinct foraging, reproductive and habitat-use guilds; Distinct physiological tolerances and ecomorphologies
Valley Segment Types	Hydrogeomorphic subunits of Aquatic Ecological Systems. Aggregates of stream reaches with broad similarities in fluvial processes, sediment transport, riparian vegetation, and thermal regime.	Temperature; Stream size; Gradient; Permanence of flow; Position within drainage network; Valley geomorphology	Species level patterns; Distinct foraging, reproductive and habitat-use guilds; Distinct physiological tolerances and ecomorphologies
Habitat Unit Types	Hydrogeomorphic subunits of Valley Segment Types (e.g., riffle, pool, run).	Depth; Velocity; Substrate; Position within the channel; Physical forming features	Species level patterns; Distinct foraging, reproductive and habitat-use guilds; Distinct physiological tolerances and ecomorphologies

3.3. Levels 1 – 3: Zone, Subzone, and Region

Objective

Identify land masses and groups of major drainages that contain relatively similar assemblages in terms family, species, and genetic composition due to similarities in evolutionary history.

General Description

The upper three levels of the hierarchy are largely zoogeographic strata representing geographic variation in taxonomic (family and species-level) composition of aquatic assemblages across the landscape resulting from distinct evolutionary histories (e.g., Pacific versus Atlantic drainages). For these three levels we adopted the ecological units delineated by Maxwell et al. (1995). Maxwell et al. (1995) used existing literature and data, expert opinion, and maps of North American aquatic zoogeography (primarily broad family-level patterns for fish and also unique aquatic communities) to delineate each of the geographic units in their hierarchy. More recent quantitative analyses of family-level faunal similarities for fishes conducted by Matthews (1998) provide additional empirical support for the upper levels of the Maxwell et al. (1995) hierarchy. The ecological context provided by these first three levels may seem of little value, however, such global or subcontinental perspectives are critically important for research and conservation (see pp. 261-262 in Matthews 1998). For instance, the physiographic similarities along the boundary of the Mississippi and Atlantic drainages often produce ecologically similar (i.e., ecological composition) riverine assemblages within the smaller streams draining either side of this boundary, as Angermeier and Winston (1998) and Angermeier et al. (2000) found in Virginia. However, from a taxonomic composition or phylogenetic standpoint, these ecologically similar assemblages are quite different as a result of their distinct evolutionary histories (Angermeier and Winston 1998; Angermeier et al. 2000). Such information is especially important for those states that straddle these two drainages, such as Georgia, Maryland, New York, North Carolina, Pennsylvania, Tennessee, Virginia, and West Virginia, since simple richness or diversity measures not placed within this broad ecological context would likely fail to identify, separate, and thus conserve highly distinctive components of biodiversity. The importance of this broader context also holds for those states that straddle the continental divide or any of the major drainage systems of the United States (e.g., Mississippi Drainage vs. Great Lakes or Rio Grande Drainage).

3.4. Level 4: Aquatic Subregions

Objective

Identify groups of major drainages that drain regions with similar physiographic character and contain relatively similar assemblages in terms of ecological composition (e.g., life history strategies and physiological tolerances).

General Description

Aquatic Subregions are physiographic or ecoregional substrata of Regions and thus account for differences in the ecological composition of riverine assemblages resulting from geographic variation in ecosystem structure and function (Figure 3.2). The three Aquatic Subregions that cover Missouri (i.e., Central Plains, Ozarks, and Mississippi Alluvial Basin) largely correspond with the three major aquatic faunal regions of Missouri described by Pflieger (1989). Pflieger (1989) used a species distributional limit analysis and multivariate analyses of fish community data to empirically define these three major faunal regions. We slightly modified the boundaries of Pflieger's faunal regions to ensure that the boundaries between Subregions followed major drainage divides in order to account for drainage-specific evolutionary histories in succeeding levels of the hierarchy. Subsequent studies examining macroinvertebrate assemblages have provided additional empirical evidence that these Subregions are necessary strata to account for biophysical variation in Missouri's riverine ecosystems (Pflieger 1996; Rabeni et al. 1997; Rabeni and Doisy 2000). Each Subregion contains streams with relatively distinct structural features, functional processes, and aquatic assemblages in terms of both taxonomic and ecological composition. Detailed biophysical descriptions of each Aquatic Subregion are provided in Appendix 3.1.

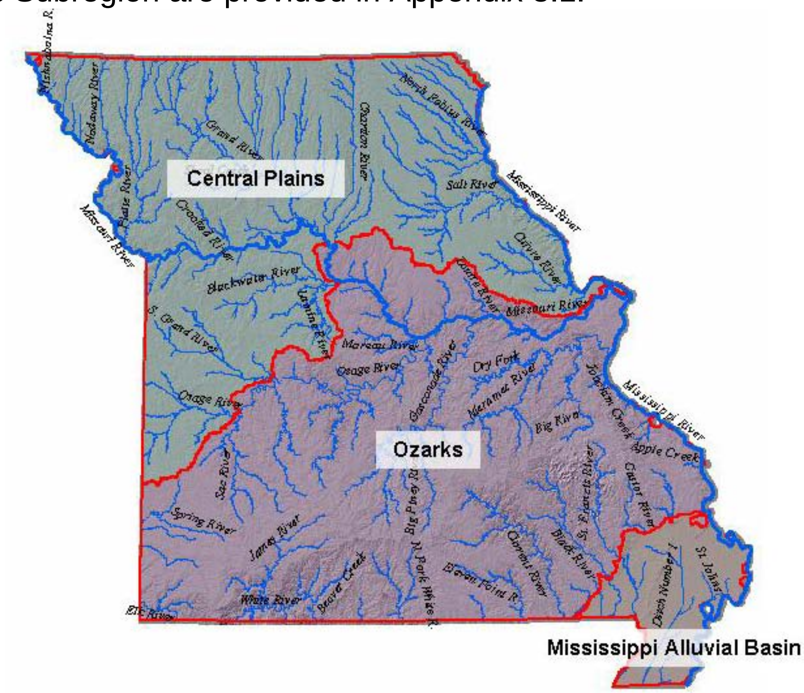


Figure 3.2. Map showing the boundaries of the three Aquatic Subregions of Missouri. Detailed descriptions of each Aquatic Subregion are provided in Appendix 3.1.

3.5. Level 5: Ecological Drainage Units

Objective

Stratify each of the Aquatic Subregions within Missouri into relatively distinct zoogeographic subunits that also fit the definition of an ecosystem.

General Description

Isolation is a key component of divergent evolutionary processes and is especially prevalent in freshwater ecosystems (Matthews 1998). For animals lacking a terrestrial life history phase, drainage boundaries serve as important isolating mechanisms, which is why each one tends to contain a relatively distinct fauna (Gilbert 1980; Pflieger 1989; Brown 1995). Embedded within Aquatic Subregions are geographic variations in taxonomic composition (species- and genetic-level) resulting from the geographically distinct evolutionary histories of the major drainages within each Subregion (Pflieger 1971; Mayden 1987; Mayden 1988; Crandall 1998; Matthews and Robison 1998). Level 5 of the hierarchy, Ecological Drainage Units (EDUs), account for these differences in taxonomic composition (Figure 3.3). EDUs are analogous to “islands” when viewed within the context of the surrounding Aquatic Subregion, which is analogous to the “sea” in which the EDUs reside. Within a given Aquatic Subregion, all of the EDUs have assemblages with relatively similar ecological composition (e.g., physiological tolerances, reproductive and foraging strategies). However, the taxonomic composition (species and phylogenetic composition) of the assemblage within any given EDU is relatively distinct due to evolutionary processes such as adaptive radiation, genetic drift, differences in colonization history, random genetic mutation, etc.

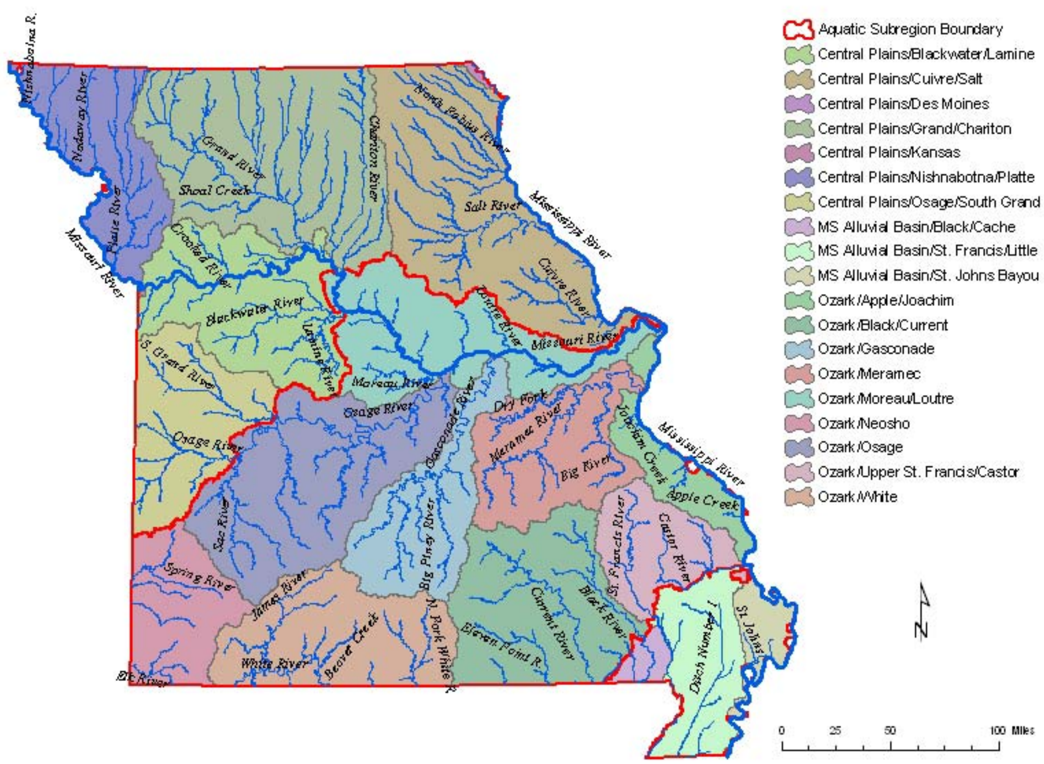


Figure 3.3. Map of Ecological Drainage Units (EDUs) for Missouri.

Methods for Classifying and Mapping EDUs

Mandatory Criteria

1. Each EDU must be fully contained within an Aquatic Subregion.
2. Each EDU must contain at least one stream classified as large river or great river (~ Strahler order $\geq$ 6).

Software Used

ArcView 3.3
ArcInfo (workstation)
SAS 8.2
PC-ORD for Windows version 4
Microsoft Excel 2000
Microsoft Access 2000

Baselayer and Source Data Used to Classify EDUs

- 1:100,000 Valley Segment Coverage attributed with stream size classes (MoRAP)
- 1:100,000 Aquatic Subregions of Missouri (MoRAP)
- 1:100,000 8-digit Hydrologic Units of Missouri (USGS)
- 1:100,000 Hydrologic Unit coverage created by intersecting Aquatic Subregions and 8-digit HUs (MoRAP)
- Existing community sampling data for fish, mussels, crayfish, and snails (Compiled by MoRAP from various sources)

General Approach

The baselayer for delineating EDUs was created by intersecting our Aquatic Subregion coverage with the USGS 8-digit Hydrologic Units (HUs) to create a new set of HUs that were fully contained within one of the three Aquatic Subregions. Next, we spatially linked thousands of existing community fish samples (presence data) to the USGS/EPA National Hydrography Dataset and the new set of HU polygons. We then quantified the prevalence of each fish species within each HU by calculating the percent occurrence of each species within a randomly selected 40-sample subset. The resulting data matrix (species percent occurrence by HU) was used as the input data for a series of multivariate analyses that statistically examined the relative similarity of fish assemblages among HUs. These analyses were performed separately for each Aquatic Subregion. Results of these analyses were used to group HUs with relatively similar in fish assemblages into an initial set of (EDUs). We then refined the boundaries of this initial set of EDUs based on a gross comparison of faunal similarities among the major drainages within each Aquatic Subregion. These analyses were based on a Jaccard Similarity Index calculated using collection data for three other taxa (crayfish, mussels, and snails).

The low number of HUs in the Mississippi Alluvial Basin (MAB) was unsuited to the multivariate analyses used for the Central Plains (CP) and Ozark (OZ) Aquatic Subregions. Consequently, we used Jacaard Similarity Indices (based on all four taxa) to group HUs with relatively similar assemblages within this Subregion. Specifically, those HUs having Jacaard Similarity coefficients greater than one standard deviation above the statewide average (i.e., Jacaard coefficient $\geq$ 67), were grouped to form the EDUs within the MAB. Using this approach a total of three EDUs were delineated for the MAB. The remainder of this section covers the more detailed methods used for delineating EDUs within the CP and the OZ.

Detailed Methods

The baselayer for delineating EDUs was created by intersecting our Aquatic Subregion coverage with the USGS 8-digit Hydrologic Units (HUs) to create a new set of HU polygons that were fully contained within one of the three Aquatic Subregions (Figure 3.4). Many of the original 8-digit HUs remained unchanged after this procedure since they were already fully contained within one of the three Subregions. However, those HUs that straddled the boundaries between any of the three Aquatic Subregions were split into two separate HUs.

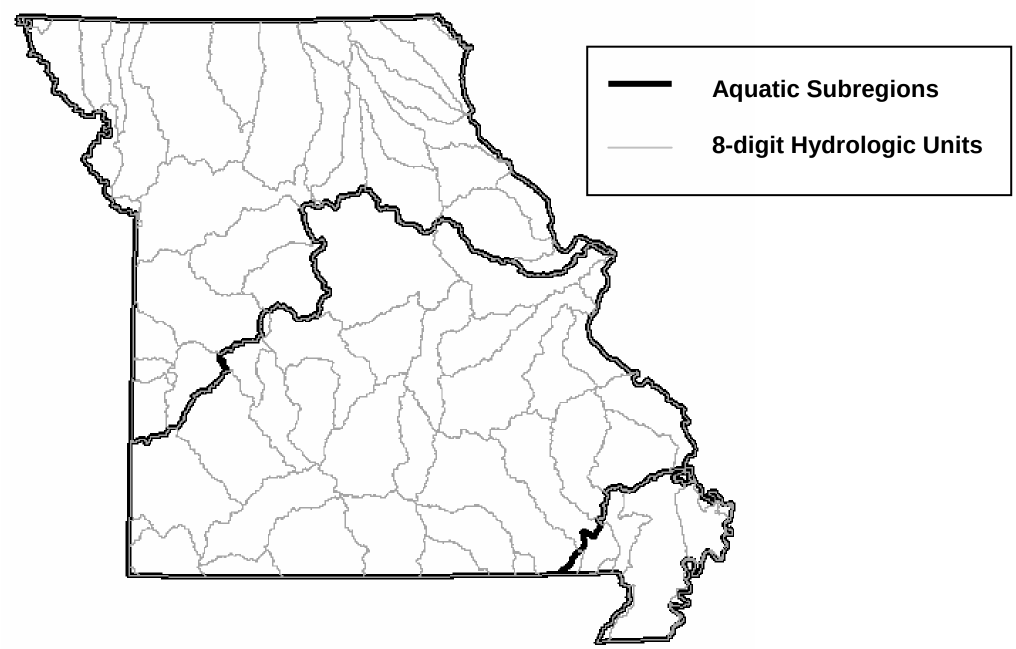


Figure 3.4. Map showing boundaries of Aquatic Subregions and USGS 8-digit Hydrologic Units (HU). These two coverages were intersected to create a new HU coverage that served as the baselayer for delineating EDUs in Missouri.

Next, we spatially linked 3,723 existing community fish samples (presence data) to the USGS/EPA National Hydrography Dataset and the newly created HU polygons. The number of samples within an HU was not evenly distributed among all of the HUs, which could have significantly distorted any assessments of faunal similarity among HUs (Figure 3.5). For instance, (dis)similarties among an HU with 50 samples to one with 250 samples may be more related to differences in sampling effort rather than actual differences in fish assemblage composition. Consequently, we carried out a preliminary set of analyses to determine the subsample size needed to maximize species capture and minimize the inequality of sample sizes among HUs. Two approaches were used to evaluate the appropriate subsample size. First, species-area curves were constructed in PC-ORD for the CP and OZ using sample sizes of 20, 30, 40, & 50 (Figures 3.6 & 3.7). These curves were constructed for four subsets of the data; all samples with all species included, only species occurring in >5% of the samples in a given HU, only species occurring in >5% of all HUs, and only species occurring in >5% of samples of a given HU and >5% of all HUs. The same pattern was observed regardless of which subset was used, although the more restrictive the subset, the more pronounced the pattern. In addition, Figures 3.6 & 3.7 show jack-knife estimates for the first two subsets of data. Collectively, these results indicate that after a given sample size, roughly 40 for either the Ozarks or CP, species richness tends to level out. Only native species were included in these and all subsequent analyses.

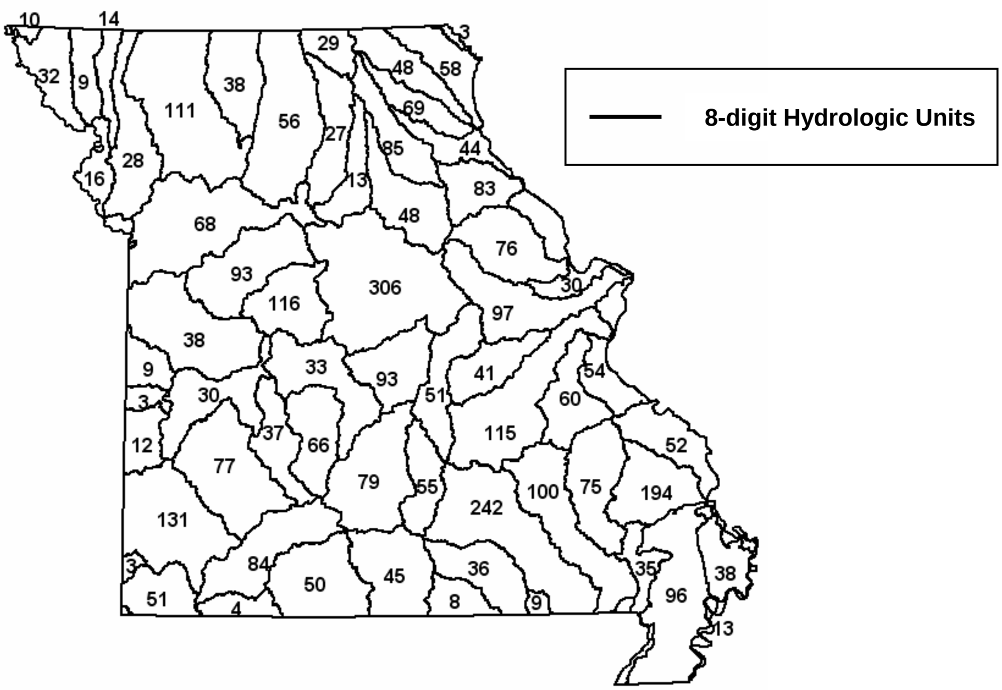


Figure 3.5. Map showing the number of fish community samples for each HU.

For the second approach we plotted the variance of the simulated mean of species richness per sample against the number of samples for all HUs within the CP and OZ. In other words, we calculated the mean variance of species richness given random selections (1000 times with replacement) of samples from the pool of available samples. This was conducted for sample sizes ranging from 20 to 50, increasing by increments of 2. The resulting plot shows the expected decline in variance with increasing sample size, with a marked drop in the 30-40 sample range (Figure 3.8). Based on the results of these two analyses we determined that at least 40 samples should be randomly selected from each HU in order to standardize sampling effort among units. Not all of the HUs contained 40 samples (see Figure 3.5) so we modified the existing HU coverage by merging HUs with less than 40 samples with the most logical adjoining HU in order to create the final baselayer for delineating EDUs (Figure 3.9).

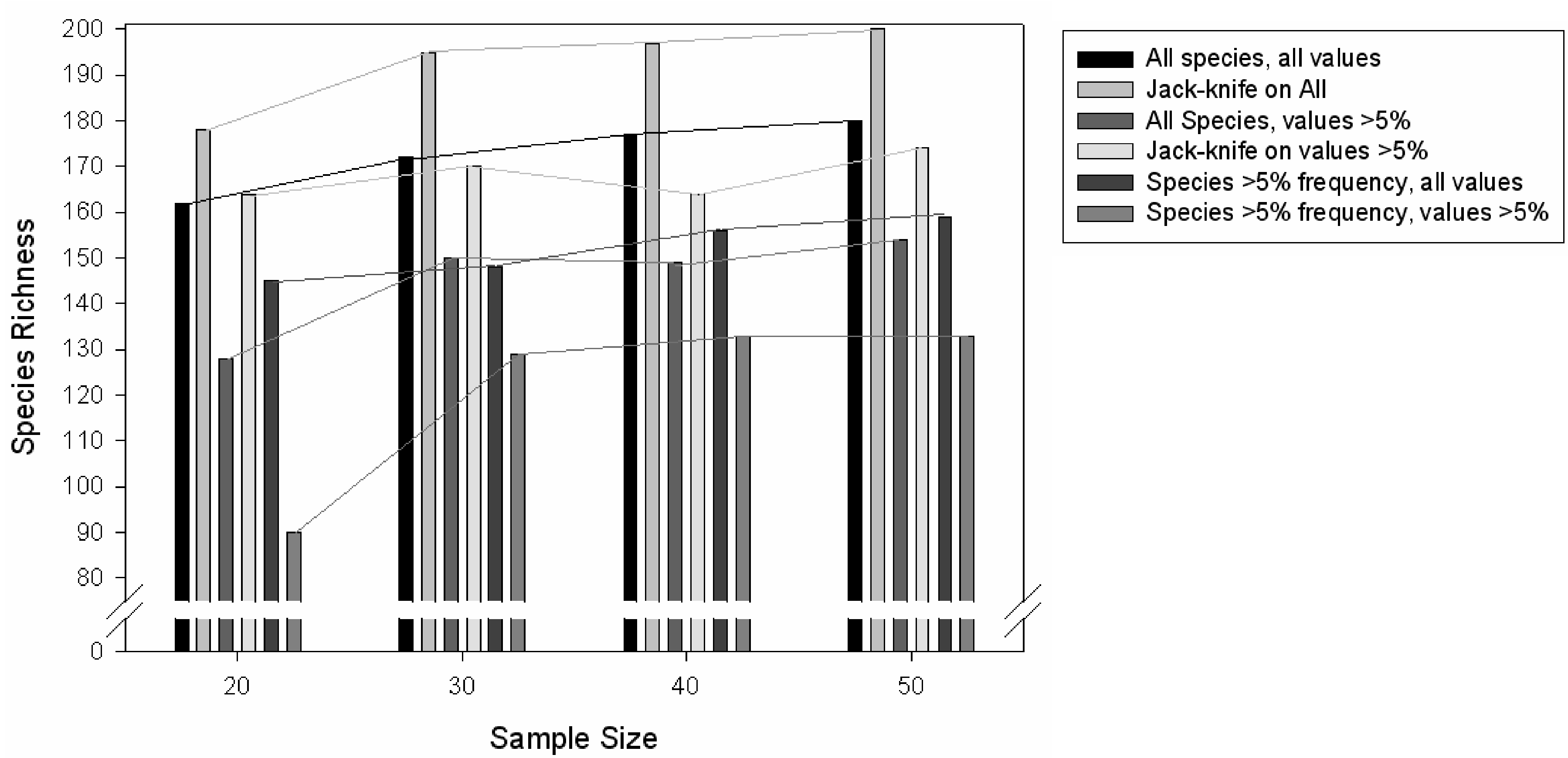


Figure 3.6. Histogram showing the average number of species captured within HUs in the Ozarks for three different samples sizes and jack-knife estimates of species richness for each sample size. See text for explanation of the different input datasets that were used to generate this histogram.

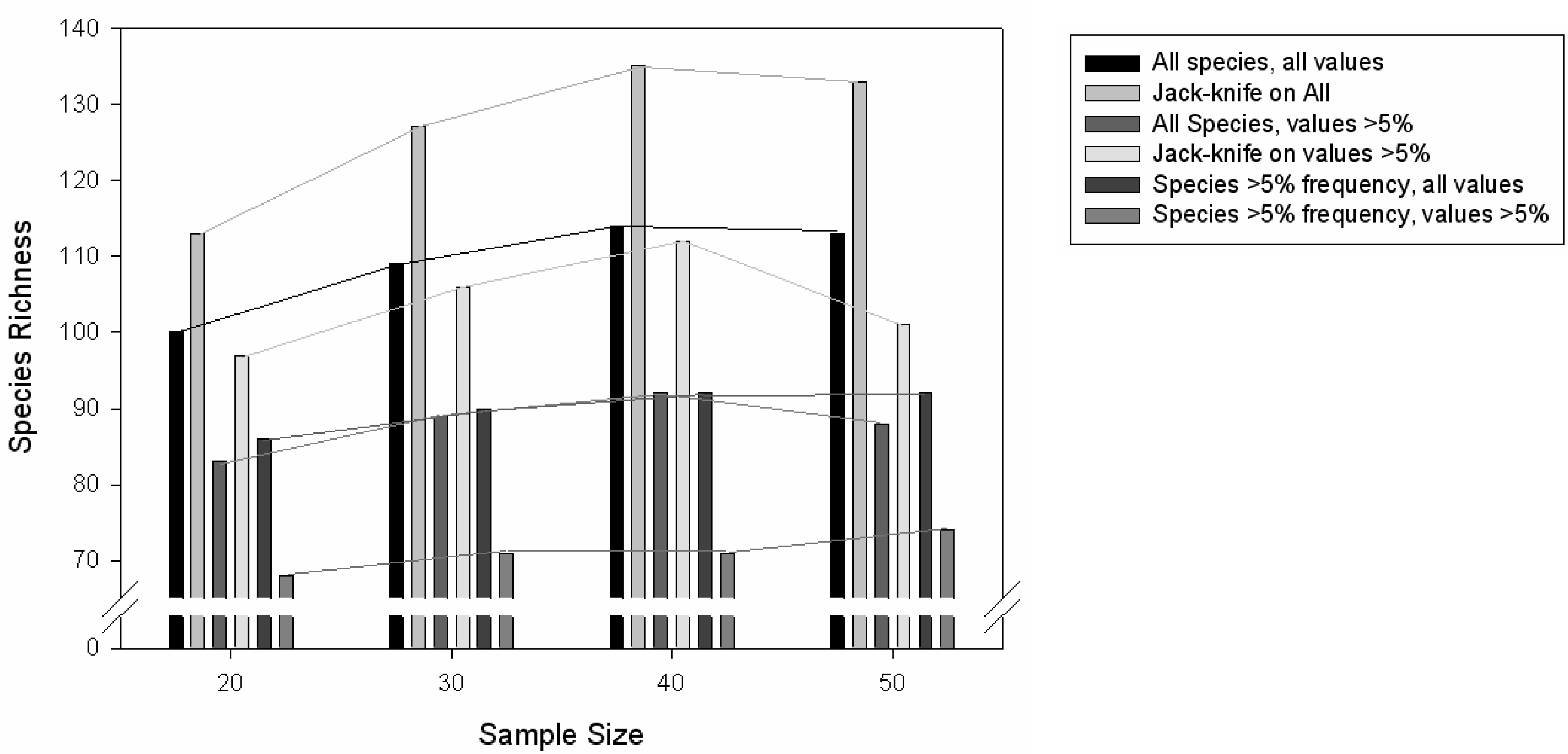


Figure 3.7. Histogram showing the average number of species captured within HUs in the Central Plains for three different samples sizes and jack-knife estimates of species richness for each sample size. See text for explanation of the different input datasets that were used to generate this histogram.

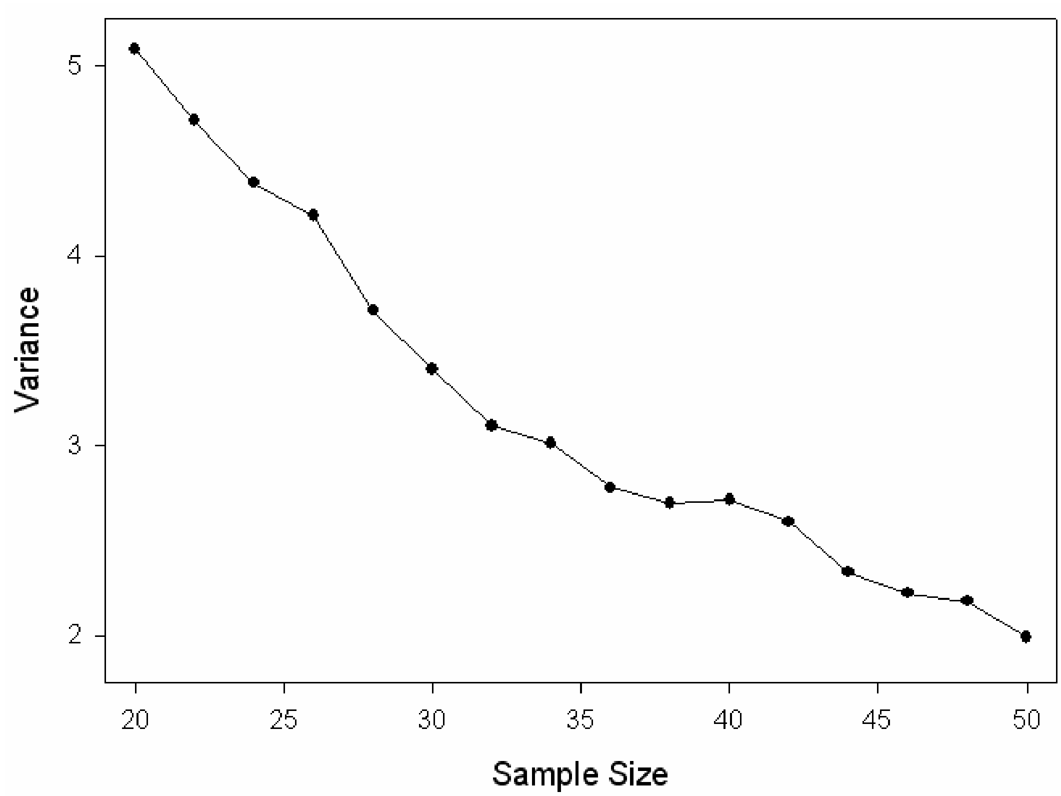


Figure 3.8. A plot of variance of the simulated mean of species richness per sample against the number of samples for all HUs within the Central Plains and the Ozarks. Specifically, this plots shows the mean variance of species richness given random selections (1000 times with replacement) of samples from the pool of available samples. Variance in species richness was calculated for sample sizes ranging from 20 to 50, increasing by increments of 2.

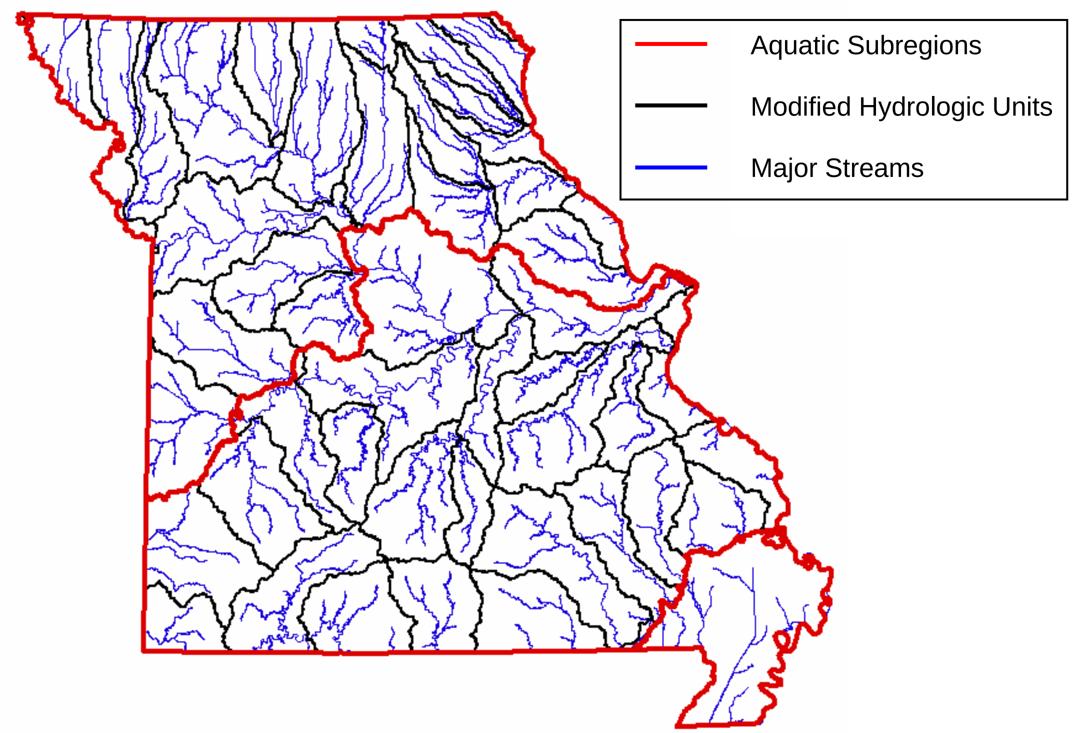


Figure 3.9. Final set of Hydrologic Units that was used as the baselayer for empirically generating Ecological Drainage Units for Missouri.

Fish assemblages can vary tremendously with stream size (Matthews 1986) and the fish community samples contained within the final set of HUs were by no means evenly distributed with regard to this parameter. Consequently, we also had to account for these sampling "biases" since they could severely affect the results of our analyses. For instance, it is possible that a randomly selected, 40-sample, subset from one HU could be comprised of mostly small and large river samples, while the subset from another HU contained mostly headwater and creek samples. In such a scenario, these two HUs, no matter how similar their overall fish assemblages really were, would almost certainly appear to be quite different based on species prevalence statistics. To account for this potential problem, we randomly selected 20 headwater/creek samples and 20 small river/large river samples from each HU in order to generate our overall random selection of 40 samples for each HU. We then created two separate data matrices, one for the OZ and one for the CP. These matrices had a column for each native species known to occur within the given Aquatic Subregion and a row for each HU. For each HU we then calculated the percentage of the 40 randomly selected samples that each native species was found in. These matrices illustrate the prevalence of each species throughout each HU, within a given Subregion. For instance, a species found in 20 of the 40 randomly selected samples within a given HU would be given a value of 50 for that HU because it was found in 50% of the samples, while a species found in only 5 of the samples would be given a value of 12.5. This random selection process and matrix construction was done four times for each Subregion, with each of the matrices analyzed separately in order to examine consistency in results.

Datasets were analyzed using a two-dimensional analysis with Nonmetric Multidimensional Scaling (NMS in PC-ORD ver. 4.0). In addition, both Principal Components Analysis (PCA) and hierarchical, agglomerative, polythetic cluster analysis were used to examine the data. These analyses were used to delineate an initial draft set of EDUs within the CP and OZ. Once a final set of EDUs was established, we used a post hoc multi-response permutation procedure (MRPP) to evaluate the statistical similarity of HUs within and among EDUs.

Prior to analysis, datasets were evaluated for the kinds of heterogeneity that can cause problems with some ordination techniques. Based on the large percentage of zero values in the matrices (63% Ozarks and 60% CP), the somewhat above optimal values for beta diversity (2.8 and 2.5) and relatively high mean skewness (2.5 and 2.2), particularly mean kurtosis (8.3 and 6.9) and coefficient of variation across cells for species (148% and 174%), methods using the Sorenson distance measure were chosen for final interpretation. Although such high coefficients of variation normally suggest some form of data transformation, in this case transformations were not considered appropriate given that the data were already relativized (percentage) values. Deletion of rare species (e.g., those occurring in 5 or fewer HUs or those occurring in less than 5 samples in a particular HU) had no noticeable effect on the ordination results, and hence all native species were included in the final analyses. Neither dataset had excessively strong outliers.

Finally, we used collection records for three taxa (crayfish, mussels, and snails) to more generally examine the faunal similarities among HUs within each Aquatic Subregion. A similarity matrix based on the Jaccard Similarity Index was constructed for the CP and OZ. HUs that were grouped in the draft set of EDUs, based on the multivariate analyses of fish community data, but had Jaccard Similarities less than one standard deviation below the overall statewide average ($\leq$ 60) were subdivided. Consequently, the analyses we used to delineate EDUs for Missouri are largely based on comparisons of fish community composition, however, they also take into consideration other taxa that tend to have more limited dispersal capabilities.

Results and Discussion

All CP ordinations were essentially identical except for rotation. Ordinations for the OZ were also very consistent, with three of the four analyses showing essentially the same pattern. Ordinations shown in this report were chosen on the basis of clarity (i.e., we selected relatively square ordinations that visually demonstrated the conclusions arrived at through both ordination and cluster analysis). In all cases, results from the PCA provided a similar solution to those obtained with the other analyses, verifying the generally strong pattern in these datasets. We used the ordination plots and clustering dendograms to group HUs with relatively similar fish assemblages into a draft set of EDUs. For both Aquatic Subregions, results of the MRPP analyses suggested that HUs belonging to the same EDU were more similar to one another than to HUs belonging to other EDUs ($p<0.0001$; in both cases).

Based on the collective results of our analyses we delineated five distinct EDUs for the CP and eight for the OZ (Figures 3.10 and 3.11). Separation and clustering of HUs on the ordination plots for the CP was quite apparent with the spatial pattern on the plot generally reflecting a north/south gradient and also an east/west gradient for the more northern drainages. No refinements to these boundaries were deemed necessary based on the Jaccard Similarities for the other three taxa. There was also good separation and clustering of HUs on the ordination plots for the OZ except for the HU representing the Little River watershed (Sub31 on Figure 3.11), which is somewhat isolated from all other HUs on the plot. Further examination of the data revealed that the relatively high prevalence of a single species within the larger rivers of this HU, the bowfin (*Amia calva*), which is rarely found in OZ streams, was responsible for its isolation from all other HUs. Other than the bowfin, this unit has prevalence values almost identical to the nearest HU (St. Francis River) on the ordination plot. We therefore grouped the Little River and St. Francis Rivers to form the St. Francis EDU, because all of the collections containing bowfin in the Little River watershed were located at the boundary of the OZ and MAB Subregions, suggesting that these were most likely stray occurrences from populations located in the MAB Subregion where this species is quite prevalent.

Like the CP, some interesting patterns were revealed on the ordination plots for the Ozarks (see Figure 3.11). HUs containing mixed complexes of fish species characteristic of both the CP and the OZ (e.g, Spring, Elk, Moreau/Perche) plotted near the top of the ordination. The cluster of HUs in the center of the plot represents subdrainages within the north flowing watersheds draining the OZ (i.e., Osage, Gasconade, and Meramec) while HUs on the bottom of the plot all fall within the major south flowing drainages (i.e., White, Current, Black, St. Francis, Little). This plot also reveals an east/west gradient, albeit with some minor deviations from this pattern.

When we examined the faunal similarities among the major drainages of the OZ, using the crayfish, mussel, and snail data, we found that one revision to this draft set of EDUs was necessary. The multivariate analyses of the fish data suggest that the Gasconade and Lower Osage River drainages should be grouped into a single EDU, which would directly correspond to one of Pflieger's faunal divisions (Pflieger 1989). This is not surprising considering that the Jaccard Similarity for these drainages is 86, when only fish are included in the calculation. However, the Jaccard Similarity drops to only 56 when crayfish, mussel, and snail species are used to calculate the index. Based on these results we believed it was necessary to break these two drainages into two distinct units since this value is less than one standard deviation below the overall average Jaccard Similarity value. Therefore, with this revision a total of nine EDUs were delineated for the OZ (Figure 3.12).

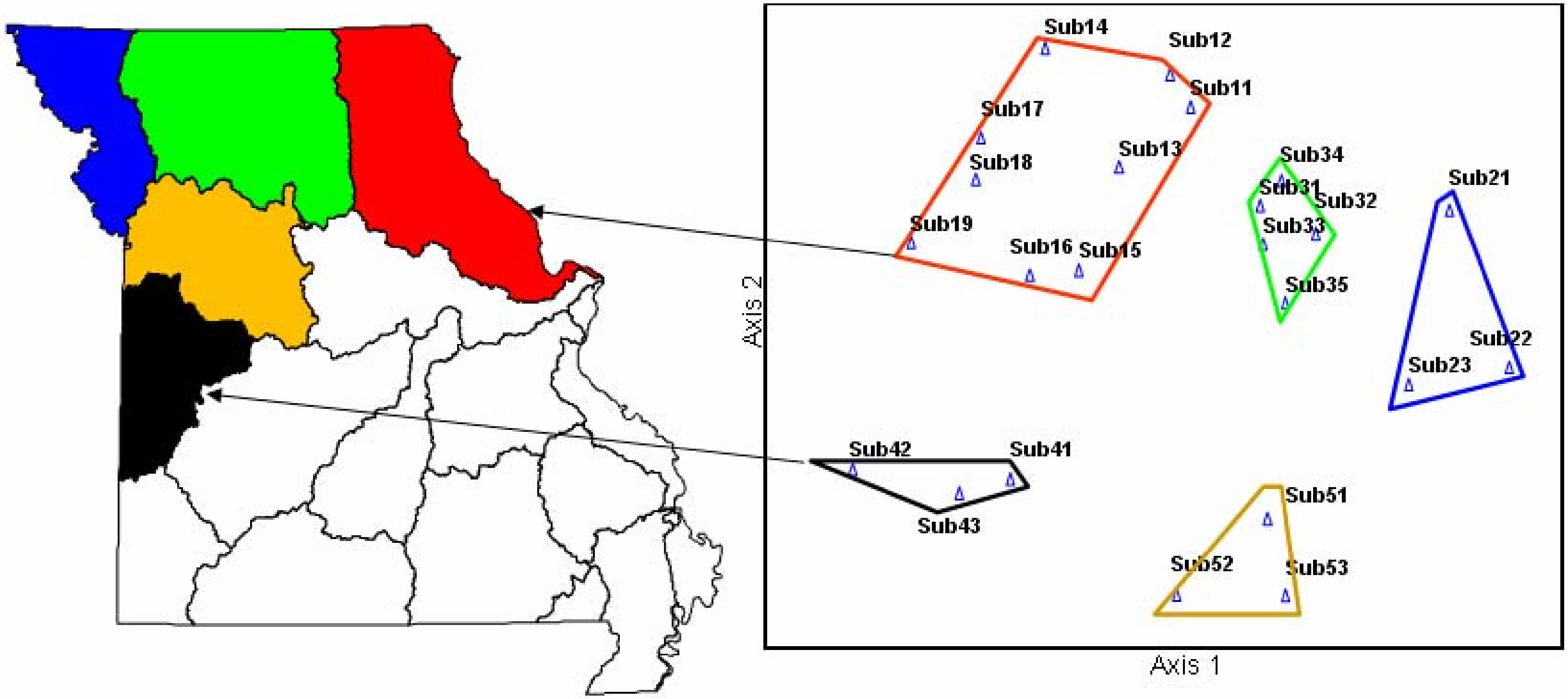


Figure 3.10. Ordination plot showing the results of an Nonmetric Multidimensional Scaling analysis performed on fish prevalence statistics for Hydrologic Units (Sub## in plot) within the Central Plains Aquatic Subregion. The color of the boxes enveloping HUs on the plot correspond with the colors of the EDUs that were generated by grouping each respective set of HUs.

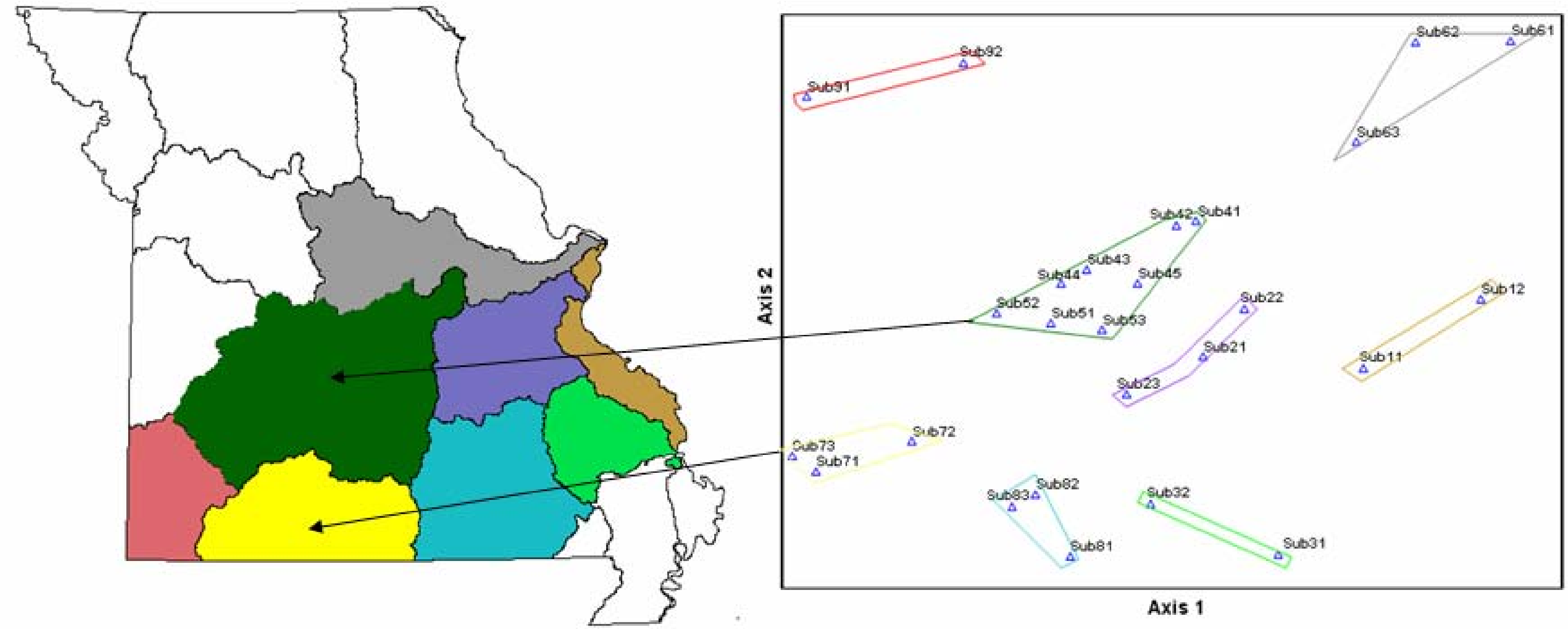


Figure 3.11. Ordination plot showing the results of an Nonmetric Multidimensional Scaling analysis performed on fish prevalence statistics for Hydrologic Units (Sub## in plot) within the Ozark Aquatic Subregion. The color of the boxes enveloping HUs on the plot correspond with the colors of the EDUs that were generated by grouping each respective set of HUs.

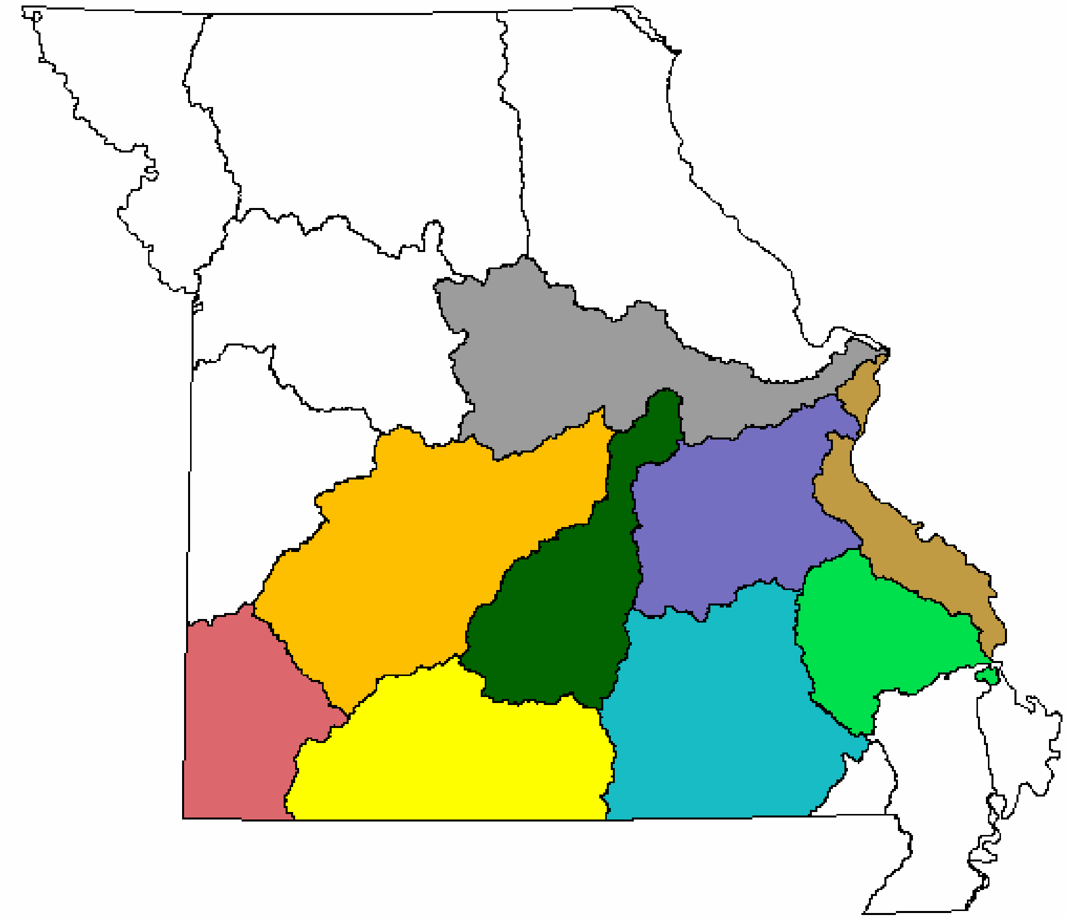


Figure 3.12. Map showing the final set of nine EDUs delineated for the Ozark Aquatic Subregion.

Based on the above analyses the total number of EDUs we delineated for Missouri is 17 (5 in CP, 9 in Ozarks, and 3 in MAB) (Figure 3.13). However, it must also be pointed out that the relatively large Kansas River and Des Moines River watersheds were not included in our analyses since only tiny fractions of these watersheds fall within Missouri and we did not have enough data to examine the relative similarity of the aquatic assemblages within these watersheds to those we did include in our analyses. Based on their size and geographic location, it is quite likely that a broader regional analysis would find these watersheds, and possibly further substrata of these watersheds, to be distinct EDUs. Consequently, it is quite possible that Missouri could contain portions of as many as 19 separate EDUs.   Detailed descriptions for each EDU can be found in Appendix 3.2.

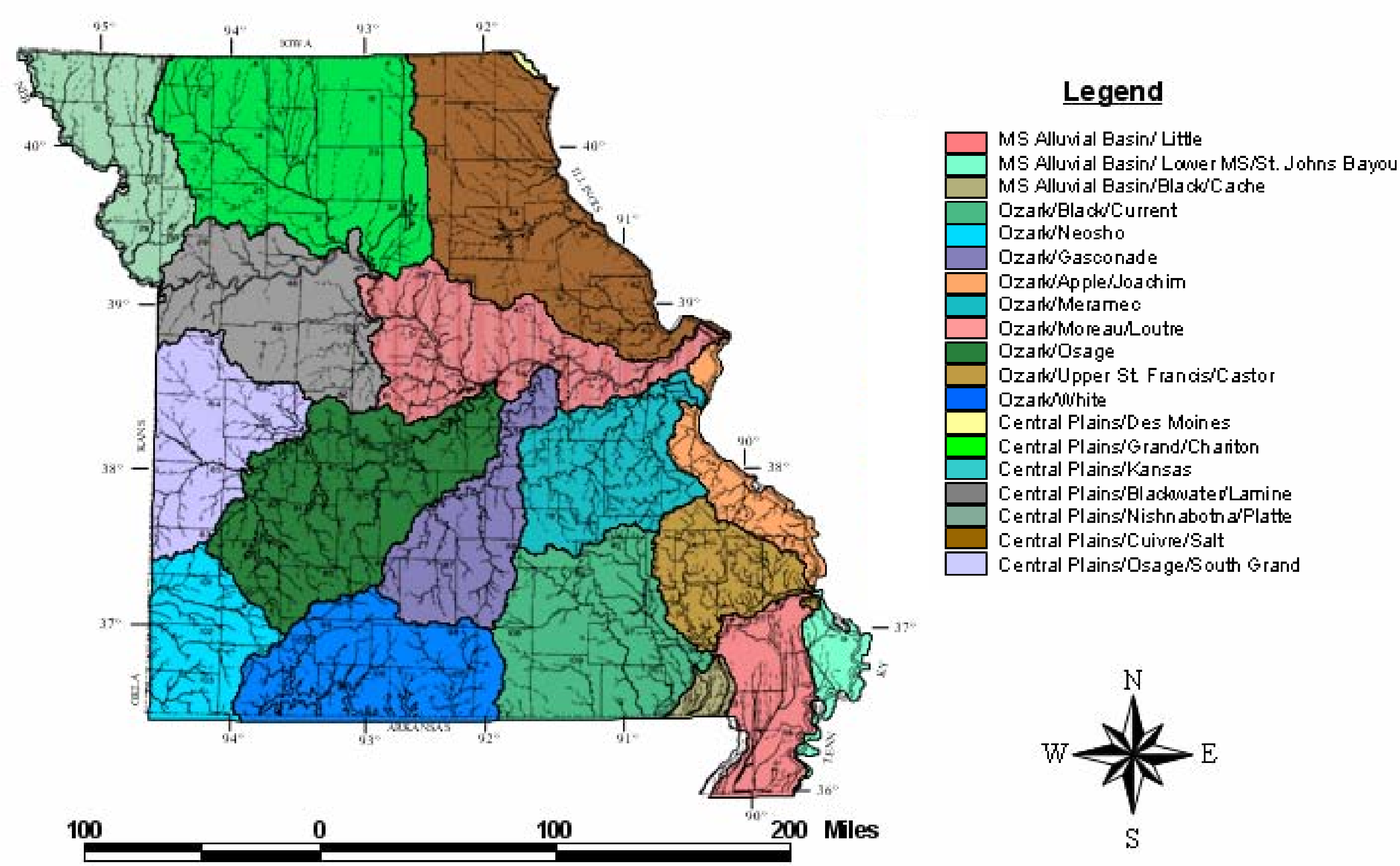


Figure 3.13. Map of Ecological Drainage Units for Missouri. Detailed descriptions for each EDU can be found in Appendix 3.2.

It is hard to say if the east/west and north/south gradients represented on both ordination plots reflect environmental gradients, geographic isolation, or relate to distances from original post-glaciation colonizing source populations in the lower Mississippi River, since all three possible explanations would fit the observed patterns. It is most likely a combination of all these factors and additional, more detailed, analyses into the phylogenetics, physiological tolerances, and life history strategies of many of these species/populations would be required to discern the relative influence of each factor. Pflieger (1971; 1989) suggests that the geographic patterns in local fish assemblage composition across Missouri are related first to geographic variations in climate, geology, soils and landform and subsequently to drainage connectivity. Matthews and Robison (1998), however, found drainage connectivity to be the principle factor associated with the similarity of fish assemblages among drainages within the Interior Highlands of Arkansas. The time over which a landscape has been free of major geologic disturbance (e.g., glaciation) likely has a major influence over whether geographic differences in assemblages are related primarily to isolation mechanisms or physiography. Landscapes like the Ozarks and Appalachian Mountains are both extremely old unglaciated landscapes where evolutionary processes have been proceeding relatively undisrupted for millions of years. In these landscapes it is not surprising to find drainage connectivity, or the distance in stream miles between any two locations, to be the principle correlate with assemblage similarity.

3.6. Level 6: Aquatic Ecological System Types (AES-Types)

Objective

Identify and map hydrologic units that are relatively similar with regard to nutrient and energy sources/dynamics, physical habitat, water chemistry, hydrologic regimes, and also contain functionally similar biological assemblages.

General Description

While Aquatic Subregions are relatively distinct in terms of their climatic, geologic, soil, landform, and stream character, they are by no means homogeneous. These finer-resolution variations in physiography also influence the ecological composition of local assemblages (Pflieger 1971; Hynes 1975; Richards et al. 1996; Panfil and Jacobson 2001; Wang et al. 2003). To account for this finer-resolution variation in ecological composition we used multivariate cluster analysis of quantitative landscape data to group small- and large-river hydrologic units into distinct Aquatic Ecological System Types (AES-Types). AES-Types represent hydrologic units, that are approximately 100 to 600 mi^2, with relatively distinct (local and overall watershed) combinations of geology, soils, landform, and groundwater influence.

AES-Types often initially generate confusion simply because the words or acronym used to name them are unfamiliar. In reality, AES-Types are just “habitat types” at a much broader scale than most aquatic ecologists are familiar with. We have no problem recognizing lake types or wetland types; AES-Types are no different except that within our classification they apply specifically to riverine ecosystems. And, just like any habitat classification, there can be multiple instances of the same habitat type. For example, a riffle is a habitat type, yet there are literally millions of individual riffles that occupy the landscape. Each riffle is a spatially distinct habitat; however, they all fall under the same habitat type with relatively similar structural features, functional processes, and ecologically-defined assemblages. The same holds true for AES-Types. Each individual AES is a spatially distinct macrohabitat, however, all individual AESs that are structurally and functionally similar fall under the same AES-Type.

One assumption for this level of the hierarchy is that under natural conditions individual AES units of the same Type will contain streams having relatively similar hydrologic regimes, physical habitat, water chemistries, energy sources, energy and sediment budgets, and ultimately aquatic assemblages. Another assumption is that each AES-Type has a relatively distinct land use potential and vulnerability to a given land use. The reason biological data were not used to empirically define and map AES-Types is that the available data was not suited to the task at hand. At this level of the hierarchy we are interested in differences in the *relative abundance* of various physiological and functional guilds, not the mere presence or absence of species and existing data are not suited to this more detailed quantification. We are also interested in defining assemblages in a pluralistic context at this level of the hierarchy. Specifically, we are trying to identify relatively distinct *complexes* of multiple local assemblages (e.g., distinct interacting complexes of headwater, creek, small, and/or large river assemblages).

Mandatory Criteria

1. Each individual Aquatic Ecological System must contain a stream classified as Small River or larger (e.g., Large or Great River).

Software Used

ArcView 3.3
ArcInfo (workstation)
SAS 8.2
Microsoft Excel 2000
SPSS for Windows ver. 12.0.1

Baselayer, Source Data, and Variables Used to Classify AES-Types

- 1:100,000 Valley Segment Coverage attributed with stream size classes
- Hydrologic units generated for all small river and larger stream segments
- STATSGO soils of Missouri (1:250,000; NRCS)
- Bedrock Geology of Missouri (1:500,000; Missouri DNR)
- Relief Grid (generated from a 30 meter DEM)
- Springs of Missouri (1:24,000/1:100,000 Missouri DNR)
- Coldwater streams of Missouri (1:24,000 Missouri Department of Conservation)

General Approach

As stated above, our objective for this level of the classification was to identify and map hydrologic units that are relatively similar with regard to nutrient and energy sources/dynamics, physical habitat, water chemistry, hydrologic regimes, and biological assemblages. Lacking sufficient field data for this broad range of factors and processes, we had to rely on a more indirect “top-down” approach that utilized surrogate landscape variables to classify distinct ecological units at this level of the classification. Specifically, for each AES polygon we quantified percentages or densities for a suite of variables (geology, soils, landform, and spring/groundwater inputs) that ultimately determine hydrologic and physicochemical conditions within stream ecosystems (Hynes 1970; Hynes 1975; Dunne and Leopold 1978; Frissell et al. 1986; Allan 1995; Richards et al. 1996; Matthews 1998). We then performed a cluster analysis on these data to group hydrologic units sharing similar percentages and densities for this suite of variables into AES-Types. We determined the number of distinct types by examining relativized overlay plots of the cubic clustering criterion, pseudo F-statistic, and the overall r-square as the number of clusters was increased (Calinski and Harabasz 1974; Sarle 1983). Plotting these criteria against the number of clusters and then determining where these three criteria are simultaneously maximized provides a good indication of the number of distinct clusters within the overall data set (Calinski and Harabasz 1974; Sarle 1983; Milligan and Cooper 1985; SAS 2001; Salvador and Chan 2003). Thirty-nine AES-Types were identified for Missouri with this method (Figure 3.14).

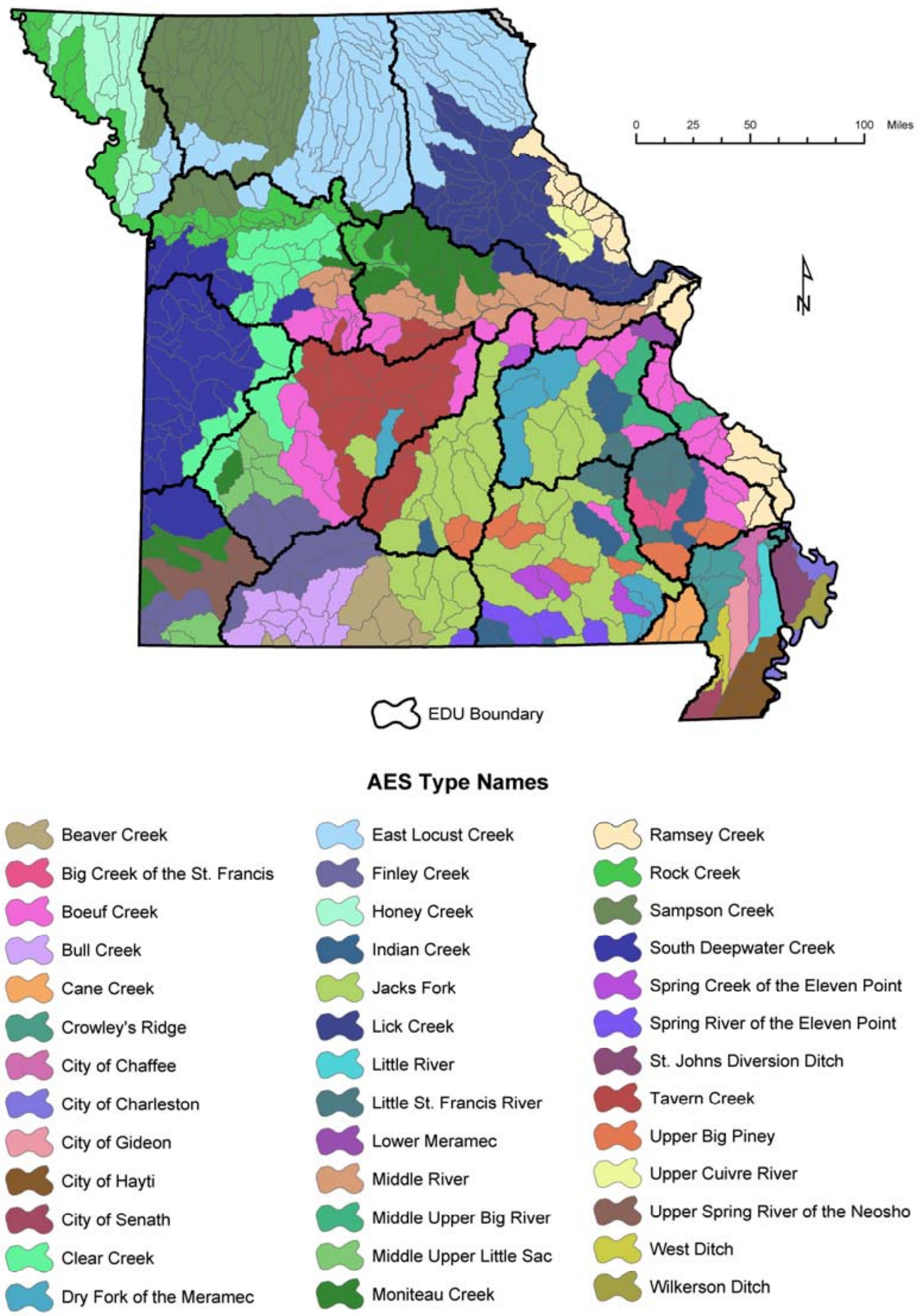


Figure 3.14. Map of the thirty-nine distinct Aquatic Ecological System Types (AES-Types) for Missouri.

Detailed Methods

Our first step in classifying Aquatic Ecological System (AES) Types involved preparing a coverage of distinct drainage polygons between all Small River and larger stream confluences. This was accomplished by taking a subset of the full drainage network contained within the 1:100,000 Valley Segment Coverage. Specifically, we removed all stream segments classified as headwater or creek from the Valley Segment Coverage and then removed all pseudo nodes to create a digital stream network that contained only streams classified as small, large, or great river (Figure 3.15). Each resulting stream segment was given a unique identifier. An AML developed by The Nature Conservancy (TNC) was used in conjunction with this reduced stream network and a 30-meter digital elevation model (DEM) to generate drainage polygons for each of the resulting stream segments (Figure 3.16). This polygon coverage was used as a template for creating a final AES coverage based on the higher resolution and more accurate 14-digit hydrologic unit coverage for Missouri. The resulting coverage served as the polygonal baselayer for calculating landscape statistics and classifying distinct AES-Types, which is discussed below. For data management purposes, each of the resulting 542 AES polygons were given a unique identifier that corresponded with the unique identifier given to the major stream segment that it contained. The only polygons that are true watersheds are those that correspond to the uppermost segments of Small Rivers, which is why we use the term hydrologic unit to describe the AES polygons.

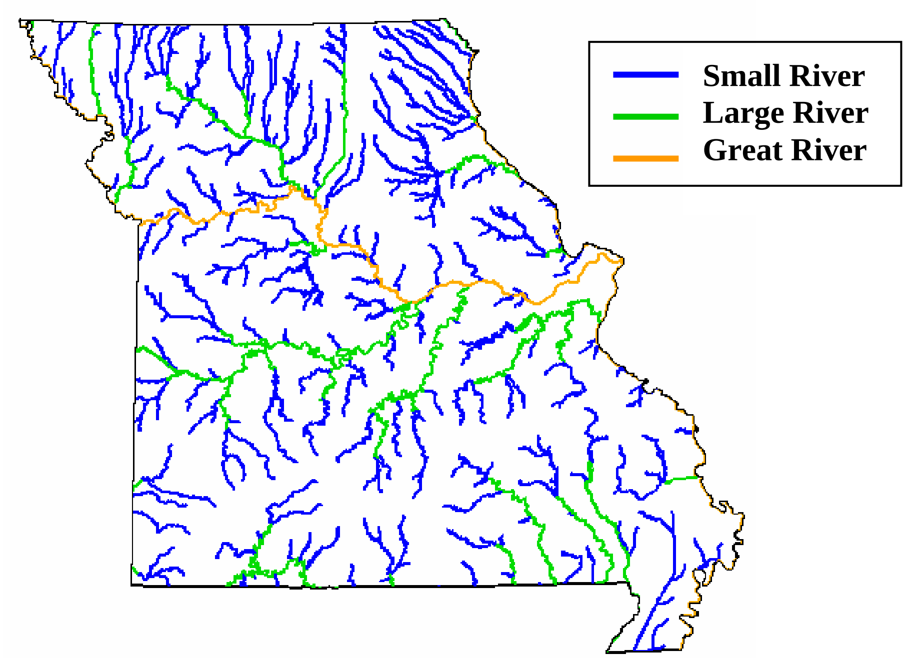


Figure 3.15. Map showing the valley segment coverage containing only streams classified as Small River or larger that was used to generate individual AES polygons.

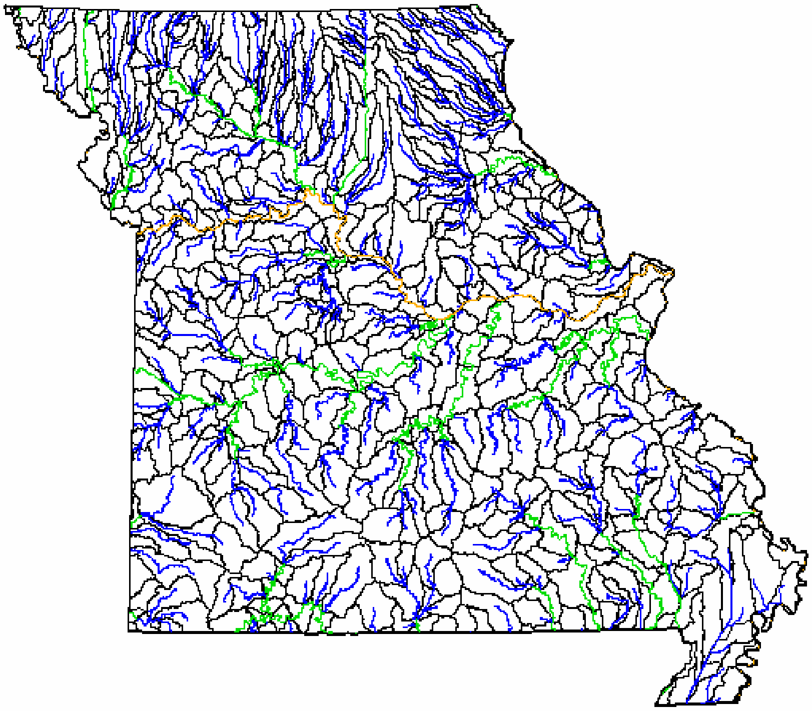


Figure 3.16. Map showing the individual AES polygons, which served as the baselayer used to classify distinct AES-Types across Missouri.

Box 3.1 Reasons Why AES-Types were only Delineated for Streams Classified as Small River or Larger

1. We wanted to represent habitat heterogeneity at broader scales and identify distinct longitudinal patterns in stream conditions. That is, we wanted to identify and classify relatively distinct complexes of headwater, creek, small, and large rivers Valley Segment Types.

2. AES-Types contain largely redundant information for headwater and creek Valley Segment Types. This is due to the fact that the variables used to classify Valley Segment Types are similar or reflect the variables used to classify AESs and there is a close correspondence between watershed conditions and local valley segment conditions in these very small streams.

3. The resulting AES polygons represent reasonably sized units (100-600 mi2) that are practical for regional planning and management.

4. The resulting AES polygons can be reasonably assumed to contain relatively distinct populations for most riverine biota (i.e., those with limited to moderate dispersal capabilities).

Input Datasets

Soils

Based on input from soil scientists, hydrologists, stream ecologists, and fluvial geomorphologists familiar with Missouri streams, we determined that Hydrologic Soil Group and soil surface texture should both be considered in the classification of AES-Types (Figure 3.17; Table 3.2). These two factors were selected because they are only moderately spatially correlated across the Missouri landscape, and yet both influence runoff and sediment production and ultimately hydrologic regimes and instream physical habitat (Richards et al. 1996; Roth et al. 1996; Allan et al. 1997; Fitzpatrick et al. 1998; Rifai et al. 2000; Panfil and Jacobson 2001). Hydrologic soil groups represent broad groups of soils having similar runoff potential under similar storm and cover conditions (USDA 2002). There are generally four hydrologic soil groups, A, B, C, and D, although some soils are placed into a combination of classes (e.g., BC), but none of these combined classes occur within Missouri. Class placement is based on the minimum annual steady ponded infiltration rate for a bare ground surface (Miller and White 1998). Soil texture refers to the percentage of sand, silt, and clay particles in a soil and classes are based upon the U.S. Department of Agriculture classification system. The seventeen soil texture classes that occur within Missouri were condensed into five general classes (Table 3.2). Soil survey data are broken in sequences and layers in order to represent the soil profile of each soil type. All of our calculations were based on values provided for Sequence 1 and Layer 1, which represent the uppermost layer of the dominant soil component within the soil profile. From the STATSGO coverage we calculated the percent area of each Hydrologic Soil Group (A, B, C, D) and five surface texture classes for each individual AES polygon and for the overall watershed draining to each individual AES polygon.

Geology

Again, based on existing research and input from a variety stream resource professionals, we determined that bedrock lithology should be considered in the classification of AES-Types. Bedrock geology influences water chemistry, flow regimes, and physical habitat (Hynes 1975; Richards et al. 1996; Roth et al. 1996; Allan et al. 1997; Matthews 1998; Fitzpatrick et al. 1998; Panfil and Jacobson 2001). The 1:500,000 statewide bedrock geologic coverage for Missouri contains both chronostratigraphic and lithostratigraphic attributes (see Figure 3.17) (MSDIS 1998). For statistical reasons we grouped the fourteen lithostratigraphic classes into six general classes (Table 2). We then calculated the percent area of each these six classes within each individual AES polygon and for the overall watershed draining to each individual AES polygon.

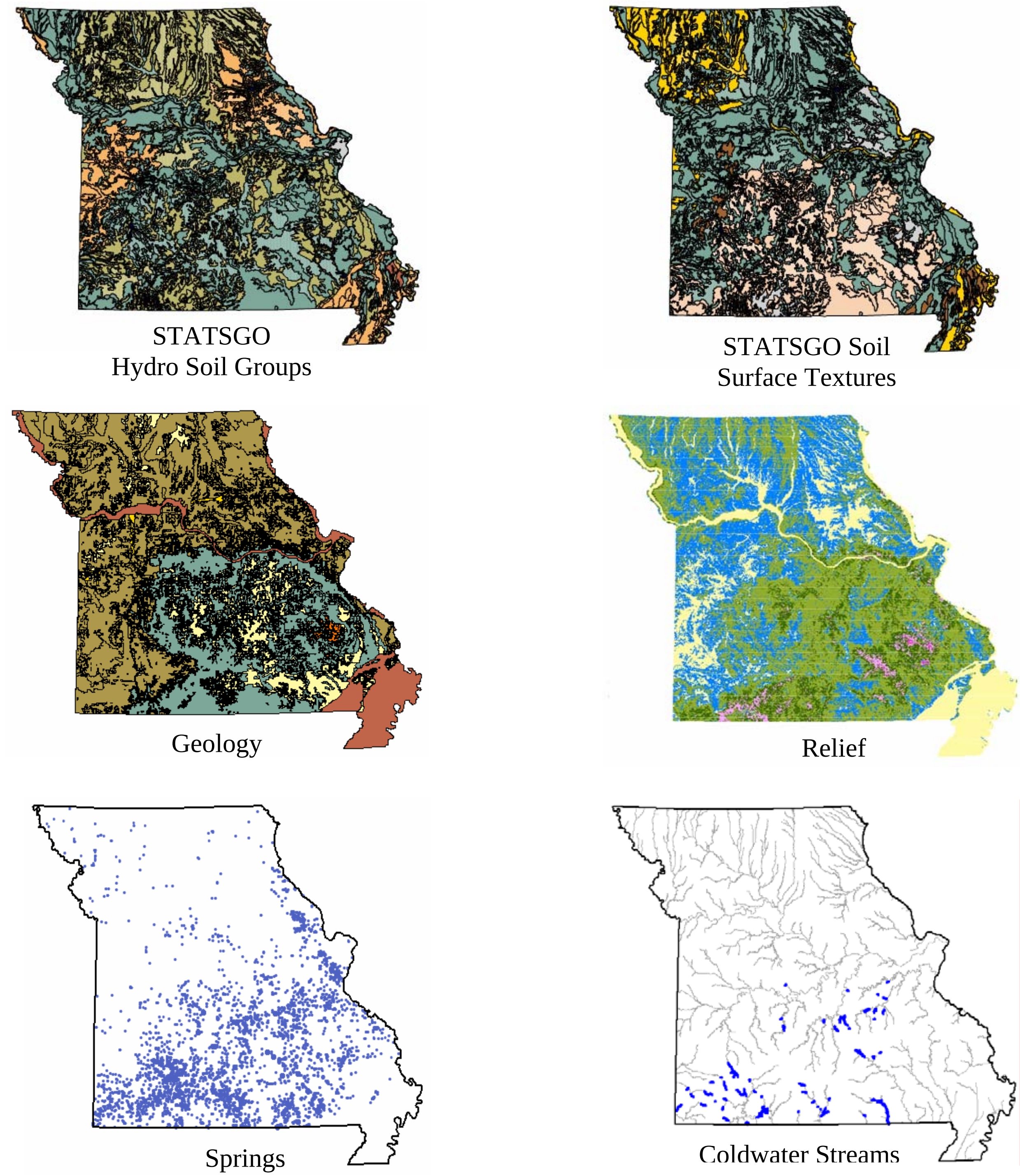


Figure 3.17. Maps showing the geospatial datasets used to generate the various landscape/landform statistics for each AES polygon, which were ultimately used in the cluster analyses to classify individual AES polygons into AES-Types. Springs and coldwater streams were used as a post cluster modifier.

Table 3.2. Hydrologic soil group and soil surface texture classes for which percent of unit area statistics were generated for each individual AES polygon and the overall watershed of each individual AES polygon.

Hydrologic Soil Groups	Original Surface Texture Classes	Condensed Surface Texture
Hydrologic Soil Group A (High Infiltration Rate)	Cherty Loam	Cherty
	Very Cherty Loam	
Hydrologic Soil Group B (Moderate Infiltration Rate)	Cherty/Silty Loam	
	Very Cherty/Silt Loam	
Hydrologic Soil Group C (Slow Infiltration Rate)	Clay Loam	Clay
	Silty Clay	
Hydrologic Soil Group D (Very Slow Infiltration Rate)	Silty Clay Loam	
	Fine Sandy Loam	Sandy
	Loamy Fine Sandy	
	Loamy sand	
	Silty Loam	Loamy
	Loam	
	Stony Loam	Stony
	Stony Silt Loam	
	Very Stony Silt Loam	
	Gravelly Fine Sandy Loam	

Table 3.3. Geologic (lithologic) classes for which percent of unit area statistics were generated for each individual AES polygon and the overall watershed of each AES polygon.

Original Geologic (lithologic) Classes	Condensed Geologic Classes
Alluvium	Alluvium
Clay	Clay
Dolomite	Dolomite
Dolomite/Limestone	
Dolomite/Shale	
Igneous	Igneous
Limestone	Limestone
Limestone/Sandstone	
Limestone/Sandstone/Shale	
Limestone/Shale	
Limestone/Shale/Sandstone	
Sandstone	Sandstone
Sandstone/Dolomite	
Sandstone/Limestone	

Relief

Landform (slope, relief, dissection) plays an important role in determining runoff and ultimately the hydrologic regime of streams (Dunne and Leopold 1978). To characterize the landform for each AES polygon we created a relief grid from the 30-meter DEM using the ArcInfo Grid command FOCALRANGE. For each cell in the input grid (30-m^2 grid cell), this command calculates the difference between the maximum and minimum elevations within a specified neighborhood surrounding each cell. We used a 1-Km^2 circle to define the neighborhood. The resulting relief grid ranged from 0-876 feet (Figure 3.17). This range was then broken into 6 relief classes (0-50, 51-100, 101-200, 201-300, 301-500, 501-700, 701-876) based upon the divisions used to create the Missouri Land Type Associations (Table 3.4; Nigh and Schroeder 2002). We then calculated the percent area of each relief class within each individual AES polygon and the overall watershed of each individual AES polygon.

Table 3.4. Relief classes for which percent of unit area statistics were generated for each individual AES polygon and its watershed.

Relief Category	Relief Range
Category 1	0 – 50 feet
Category 2	51 – 100 feet
Category 3	101 – 200 feet
Category 4	201 – 300 feet
Category 5	301 – 500 feet
Category 6	501 – 700 feet
Category 7	701 – 876 feet

Springs and Coldwater Streams

To account for significant spring and groundwater influences we used a springs of Missouri coverage developed by the Missouri Department of Natural Resource and a coldwater streams of Missouri coverage developed by the Missouri Department of Conservation (see Figure 3.17). The springs of Missouri coverage contains point locations for 4,369 springs in the state and baseflow discharge readings for 642 of these springs. These readings include most, if not all, of the springs with any significant discharge (Vineyard and Feeder 1979). Spring density was calculated for each individual AES polygon. Although spring density was also calculated for overall watershed of each AES polygon, these data were not used in the classification. The coldwater streams of Missouri coverage contains arcs representing 69 known coldwater streams in the state (i.e., rarely, if ever, have temperatures above 70 ^{o}F).

Calculating Values for Classification Variables Used in the Cluster Analysis

Percent area statistics were generated for 22 variables; 6 general geologic classes, 4 hydrologic soil groups, 5 soil surface texture classes, and 7 relief classes (Table 3.5). Percent area statistics, for each of these variables, were calculated for the both local (individual AES polygon) and overall watershed draining to each AES polygon (Figure 3.18). Consequently, for each AES polygon we generated percent area statistics for a total of 44 parameters (22 local and 22 watershed). These 44 parameters were then used as the input data for the cluster analysis in order to identify relatively distinct groupings of AES polygons. For the uppermost AES polygons, the values for the 22 local parameters were identical with the 22 watershed factors. However, for all other units the values for these two sets of parameters were different. The reason we generated both local and overall watershed statistics for each AES polygon is that significant changes in stream conditions can occur as a result of changes in local character, the issuance of a major tributary draining an entirely different landscape, or both (Figures 3.19 and 3.20).

To calculate percent area statistics for the overall watershed of each AES polygon we joined the polygonal attributes to the corresponding stream network (small rivers and larger streams) via the common identifier and subsequently traced the stream network to accumulate the total watershed area for each of the 22 variables and these values were applied to each individual AES polygon (Figure 3.21). This was accomplished using the TRACE ACCUMULATE command in ArcPlot by accumulating the area of each individual AES polygon progressively and then converting this to a percent of the entire area above the outlet of every AES polygon for each variable.

Table 3.5. Landscape variables and associated classes that were used in the cluster analyses.

Data Source:	**1:500,000 Statewide Geology**	**1:250,000 STATSGO Soils Data**		**30-m DEM**
Landscape Factor	**General Geology**	**Hydrologic Soil Group**	**Soil Surface Texture**	**Relief**
Classes	Alluvium	Group A	Clays	0-50 feet
	Clay	Group B	Cherty	51-100 feet
	Dolomite	Group C	Loams	101-200 feet
	Igneous	Group D	Sandy	201-300 feet
	Limestone		Stony	301-500 feet
	Sandstone			501-700 feet
				≥ 701 feet

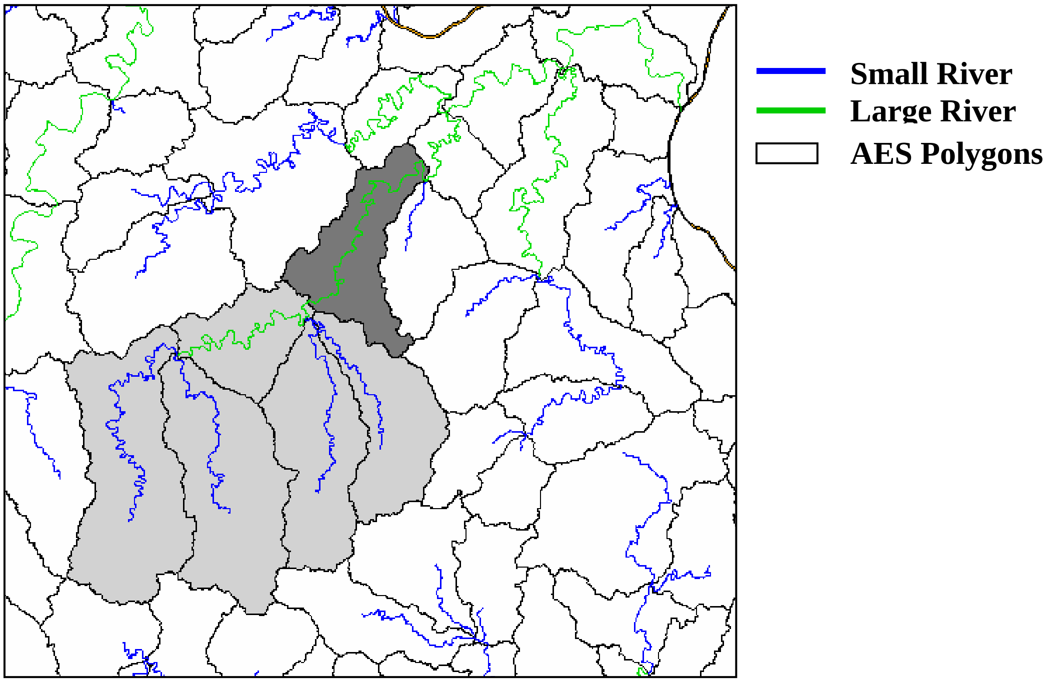


Figure 3.18. Map showing an example of the two spatial scales (local and overall watershed) at which landscape statistics were generated for each AES polygon. The dark grey polygon shows the local drainage for an individual AES polygon, while the entire shaded area represents the overall watershed for that same AES polygon. Percent area statistics were generated for both of these geographic areas for all 22 landscape variables, which resulted in a total of 44 variables used in the classification of AES-Types. This was done to account for the fact that significant changes in stream ecosystem structure and function can occur as a result of either changes in local or overall watershed conditions (see Figures 3.19 and 3.20).

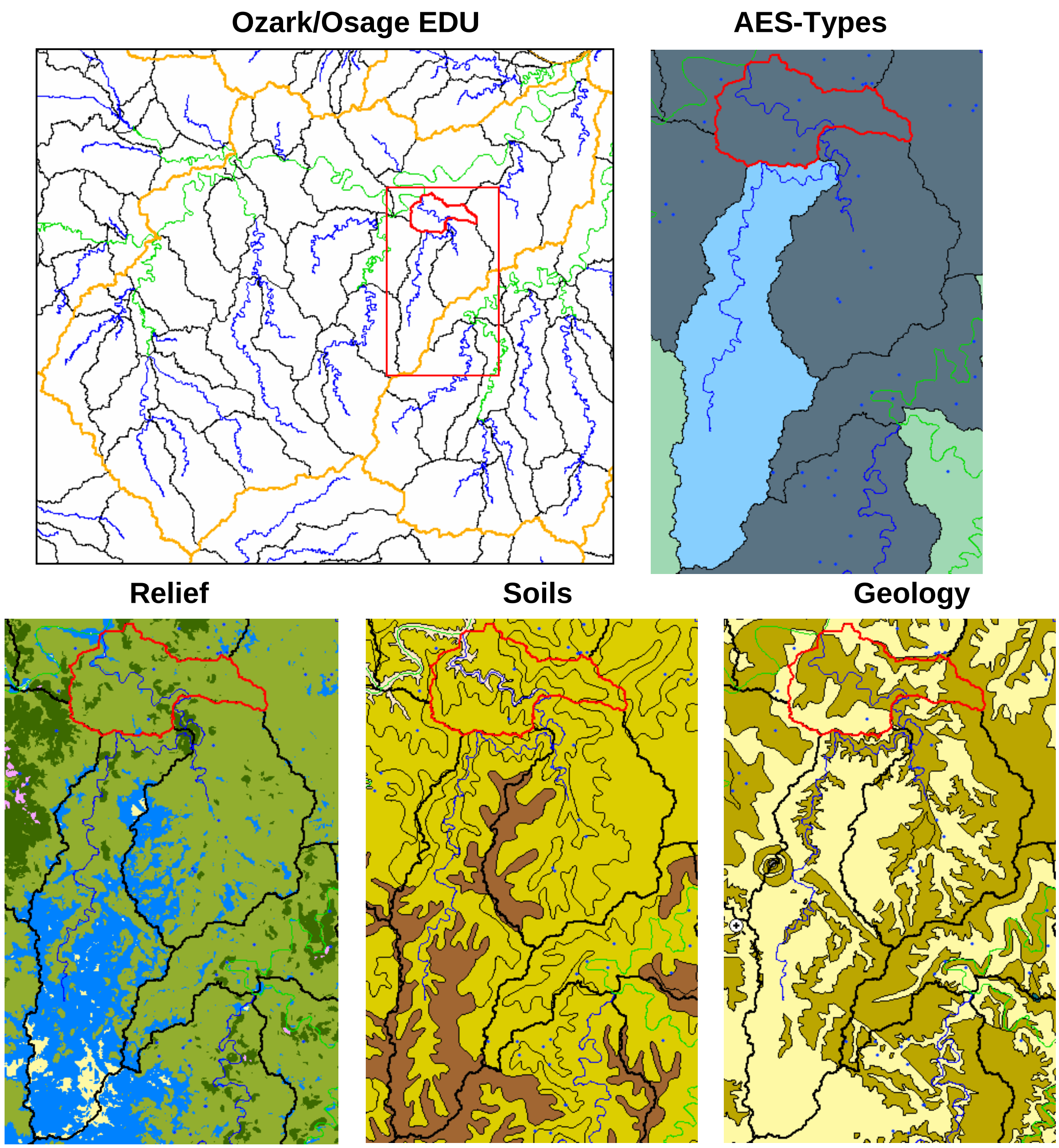


Figure 3.19. An example, within the Ozark/Osage Ecological Drainage Unit, showing how changes in overall watershed conditions resulted in changes in AES-Type designations. The AES polygon outlined in red contains the Grand Glaize River, below the confluence of the Dry Auglaize (blue) and Wet Glaize (dark grey) Creeks (see map in upper right). The Dry Auglaize watershed is characterized by lower relief (in blue and yellow) and a higher percentage of loamy soils (in brown) and sandstone (in light yellow) than Wet Glaize watershed. The overall watershed and the local drainage of the Grand Glaize River more closely resemble conditions found within the Wet Glaize watershed than that of the Dry Auglaize and is therefore classified as the same AES-Type as the polygon encompassing Wet Glaize Creek.

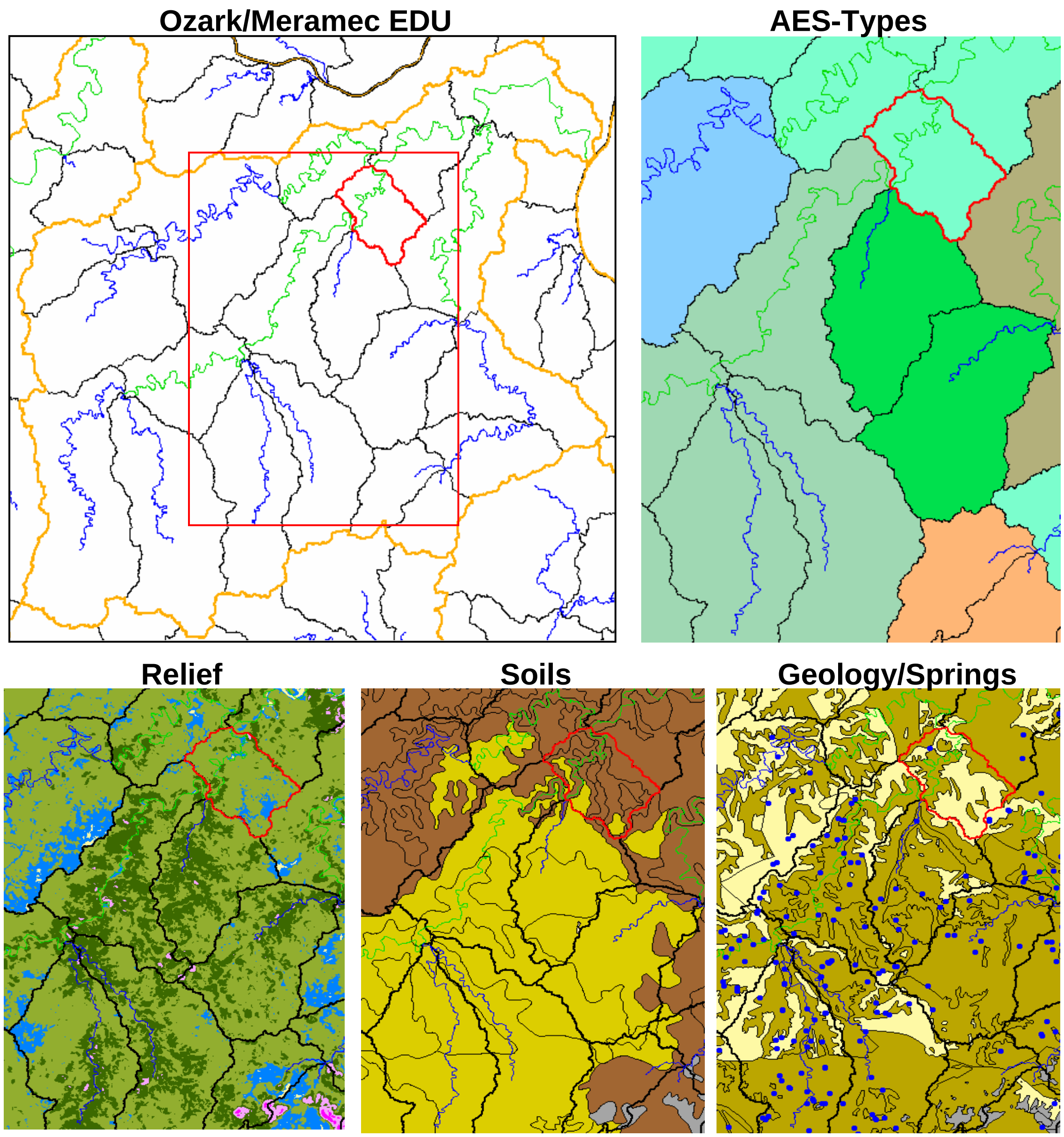


Figure 3.20. An example, within the Ozark/MeramecEcological Drainage Unit, showing how changes in local drainage conditions resulted in changes in AES-Type designations. The AES polygon outlined in red contains the lower Meramec River. At this point the river enters a region of significantly lower relief (light green and blue) that is dominated by loamy soils (brown) and sandstone bedrock (light yellow) rather than coarse-textured soils (dark yellow) and dolomite (gold) found in the upstream units. This AES polygon is also characterized by an extremely low density of springs (blue dots) relative to the upstream units.

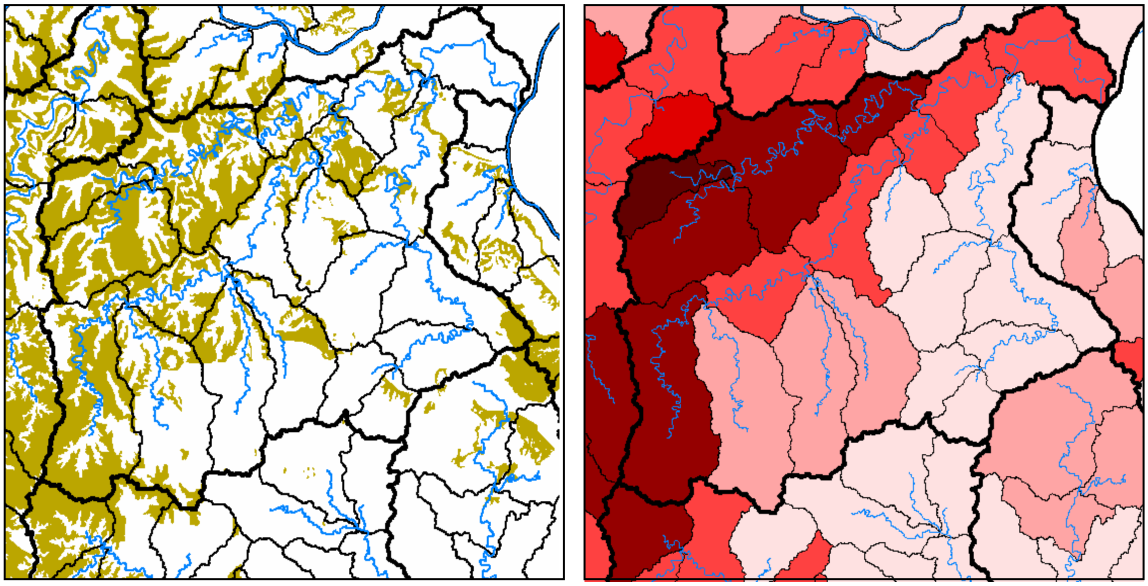


Figure 3.21. The map on the left shows the spatial distribution of sandstone bedrock (in gold) within the Meramec Ecological Drainage Unit. The map on the right shows accumulated percentage of sandstone within the overall watershed of each AES polygon. Darker colors indicate a higher percent of sandstone.

Statistical Methods

Multivariate clustering was performed with the FASTCLUS procedure in SAS 8.0.2. Cluster analysis is a multivariate analysis technique that seeks to organize information about variables so that relatively homogeneous groups, or "clusters", can be formed. The resulting clusters should be internally homogenous (members are similar to one another) and externally heterogeneous (members within one cluster are dissimilar from members of other clusters). Consequently, our objective for this analysis was to identifying AES polygons that are relatively similar with regard to the 44 parameters used as input for the cluster analysis.

Our cluster analysis was limited to the AES polygons within the Central Plains (CP) and Ozarks (OZ) of Missouri. Despite the seemingly homogenous character of the Mississippi Alluvial Basin (MAB) landscape, the ditches and few remaining natural streams and wetlands within this Aquatic Subregion vary substantially in terms of discharge, turbidity, current, substrates, aquatic vegetation and shading by riparian vegetation (Pflieger 1971). Most of this variation is associated with variations in stream size and subtle variations in soil character and elevation. In many instances, elevational differences of only a few inches will result in great differences in soil saturation characteristics and plant distribution (Brown et al. 1999). These subtle, yet important, differences in landscape character are not adequately captured with the geospatial datalayers we used to classify AESs for the CP and the OZ. Consequently, we used a geospatial coverage of the Landtype Associations delineated by Nigh and Schroeder (2002) and their source data as the geographic template for manually delineating AESs within the MAB (Figure 3.22).

Cluster analysis methods will always produce groupings, which may or may not prove useful for classifying objects of interest. If the groupings discriminate between variables not used to do the grouping (e.g., instream habitat) and those discriminations are useful, then cluster analysis is useful. Consequently, an assumption of our project is that the variables used to identify clusters (geology, soils, and landform) are significantly related to the structure and function of the stream ecosystems. With this assumption we expect streams of similar size and also both local and overall watershed geology, soils, and landform to be relatively similar with regards to water chemistry, energy dynamics, in-stream habitat, flow regimes, and resident biota.

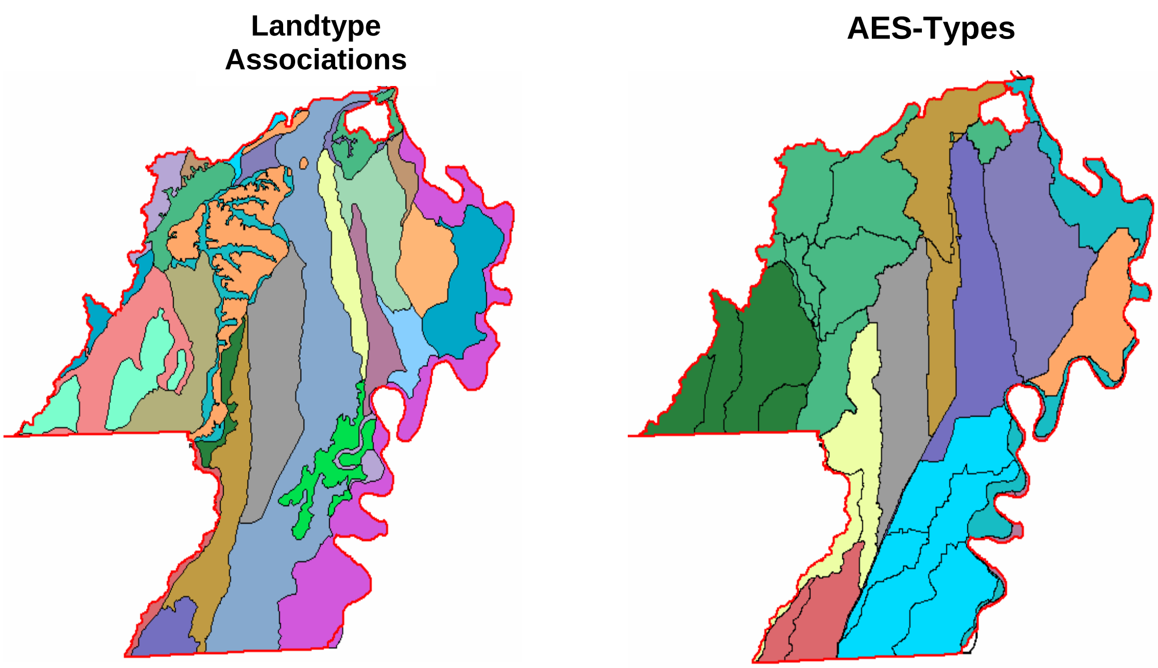


Figure 3.22. Map showing Landtype Associations (Nigh and Schroeder 2002) and AES-Types for the Mississippi Alluvial Basin Aquatic Subregion.

Identifying the appropriate number of clusters

There are no completely satisfactory methods for determining the true number of clusters for any type of cluster analysis (Everitt 1979; Bock 1985; Hartigan 1985). Ordinary significance tests, such as analysis-of-variance F-tests, are not valid for testing differences between clusters. Since clustering methods attempt to maximize the separation between clusters, the assumptions of the usual significance tests, parametric or nonparametric, are drastically violated. For example, if you take a sample of 100 or 1000 observations from a single univariate normal distribution, have PROC FASTCLUS divide it into two clusters, and perform a t-test to compare the cluster means, you usually obtain a significant P-value (SAS 2001). There are, however, various external or internal criteria that can be used to help determine the appropriate number of clusters within a particular multivariate dataset (Jongman et al. 1995). External criteria are not dependent upon the method of clustering since independent data are used to test whether or not the clustering results are meaningful. However, in our case, external data, such as species composition or abundance, water chemistry, flow regimes, or instream habitat, were not available and therefore could not be used to assess the proper number of clusters. Internal criteria are dependent upon the data used for obtaining the clusters and also the specific clustering method. Most often, two types of internal criteria are used to determine the optimum solution (Jongman et al. 1995). The first is the homogeneity of the clusters, which requires some measure of the (dis)similarity of the members of each cluster. The second is the degree of separation of the clusters, which requires some measure of the (dis)similarity of each cluster to its nearest neighbor. Typically, plots of these internal criteria against the number of clusters are used to guide the decision of how many clusters are optimal (Jongman et al. 1995; Salvador and Chan 2003).

In addition, PROC FASTCLUS provides estimates of the overall r-square, a pseudo F-statistic, and the cubic clustering criterion (Calinski and Harabasz 1974; Sarle 1983). Plotting these statistics against the number of clusters and then determining where all three are simultaneously maximized, also provides a good indication of the proper number of clusters within the overall dataset (Milligan and Cooper 1985; SAS 2001). However, caution must be used with these statistics when the discriminatory variables are correlated, which does occur in our case. It must also be emphasized that these criteria are appropriate only for compact or slightly elongated clusters, preferably clusters that are roughly multivariate normal.

We used all three of the internal criteria described above to provide insight into the proper number of clusters for each dataset. Specifically, we generated three separate diagnostic plots for the dataset.

1. Plots of the mean root-mean-square distance between observations within clusters versus the number of clusters (Figure 3.23). (Provides a means of assessing the relative homogeneity of observations within clusters as the number of clusters changes).

2. Plots of the mean distance among cluster centroids versus the number of clusters (Figure 3.24). (Provides a means of assessing the degree of separation among clusters as the total number of clusters changes).

3. Overlay plots of the overall r-square, cubic clustering criterion (CCC), and pseudo F-statistic values versus the number of clusters (Figure 3.25). (Provides a means of collectively assessing how much of the overall variance in the dataset is explained by the clusters (overall r-square), the significance/validity of the clusters against the null hypothesis of a multivariate uniform distribution (CCC), and relative significance of the differences among the cluster means (pseudo F-statistic) as the number of clusters changes).

Agreement among these various diagnostic plots, as to how many clusters actually exist within the dataset, generally provides a good indication of the number of distinct clusters (Cooper and Milligan 1984; Milligan and Cooper 1985).

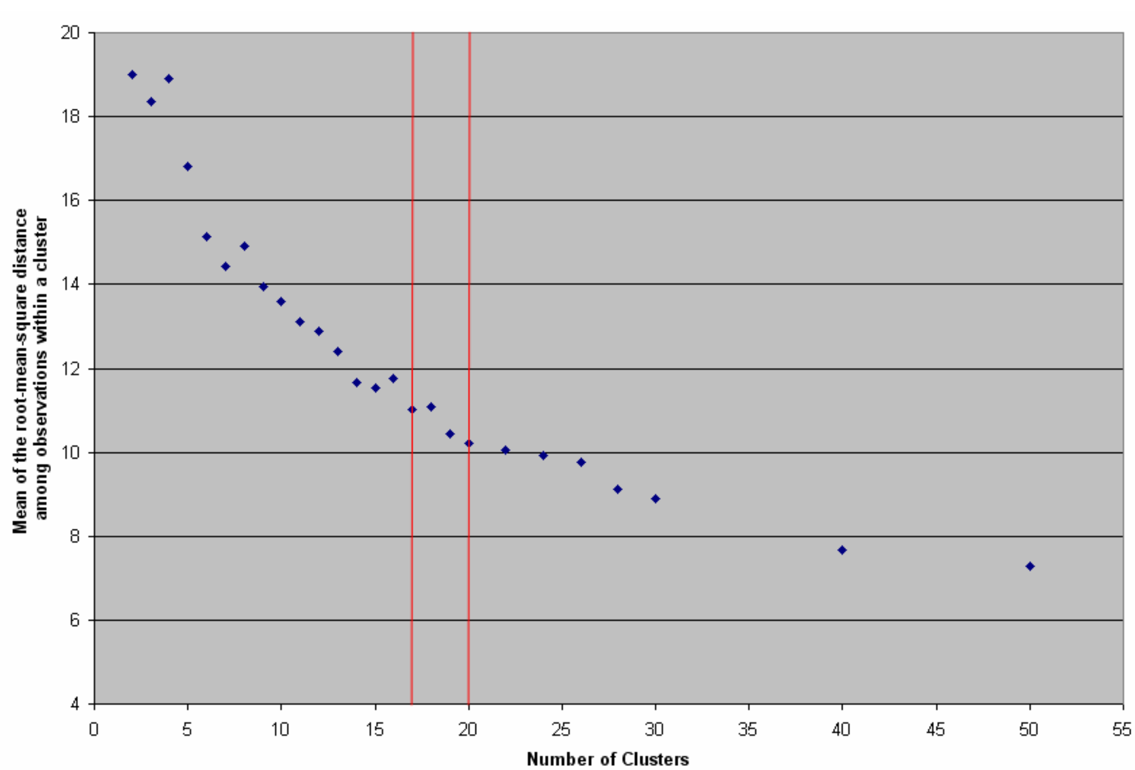


Figure 3.23. Scatter plot of the mean of the root-mean-square distance among observations within all clusters versus the number of clusters. This plot suggests that somewhere above 17 to 20 clusters (see red lines) only minimal additional variation within the set of classification variables is accounted for.

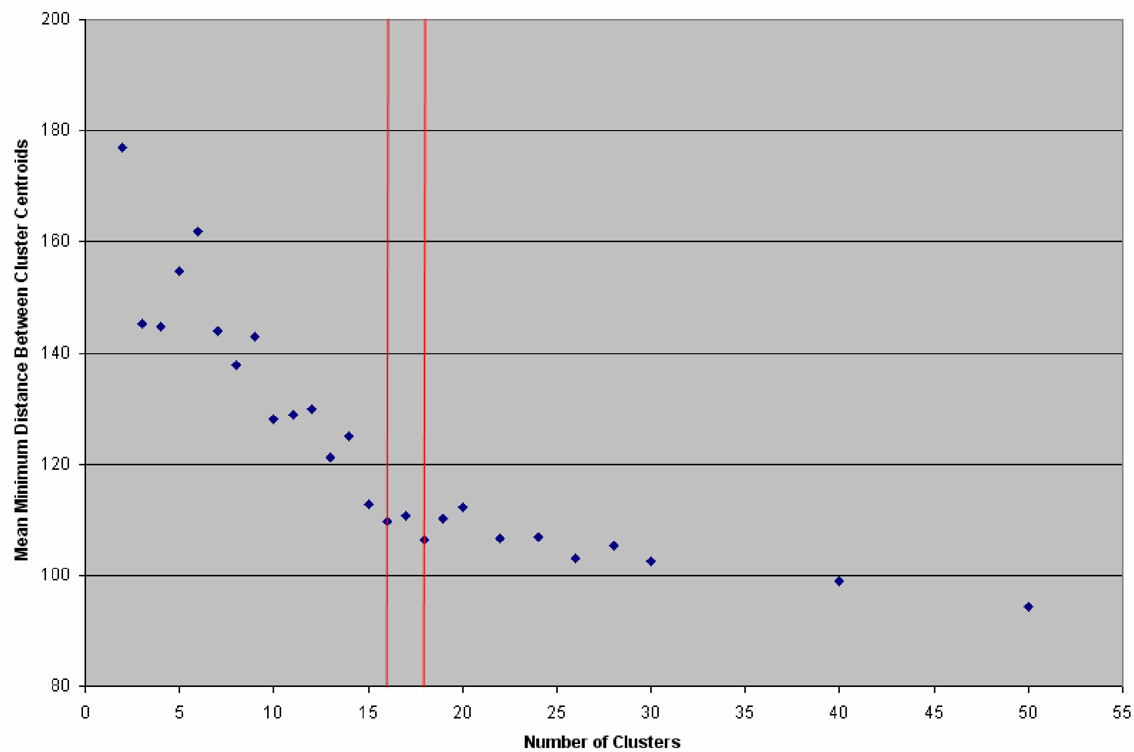


Figure 3.24. Scatter plot of the mean distance between cluster centroids versus the number of clusters. This plot suggests that somewhere above 16 to 18 clusters (see red lines) only minimal additional variation within the set of classification variables is accounted for.

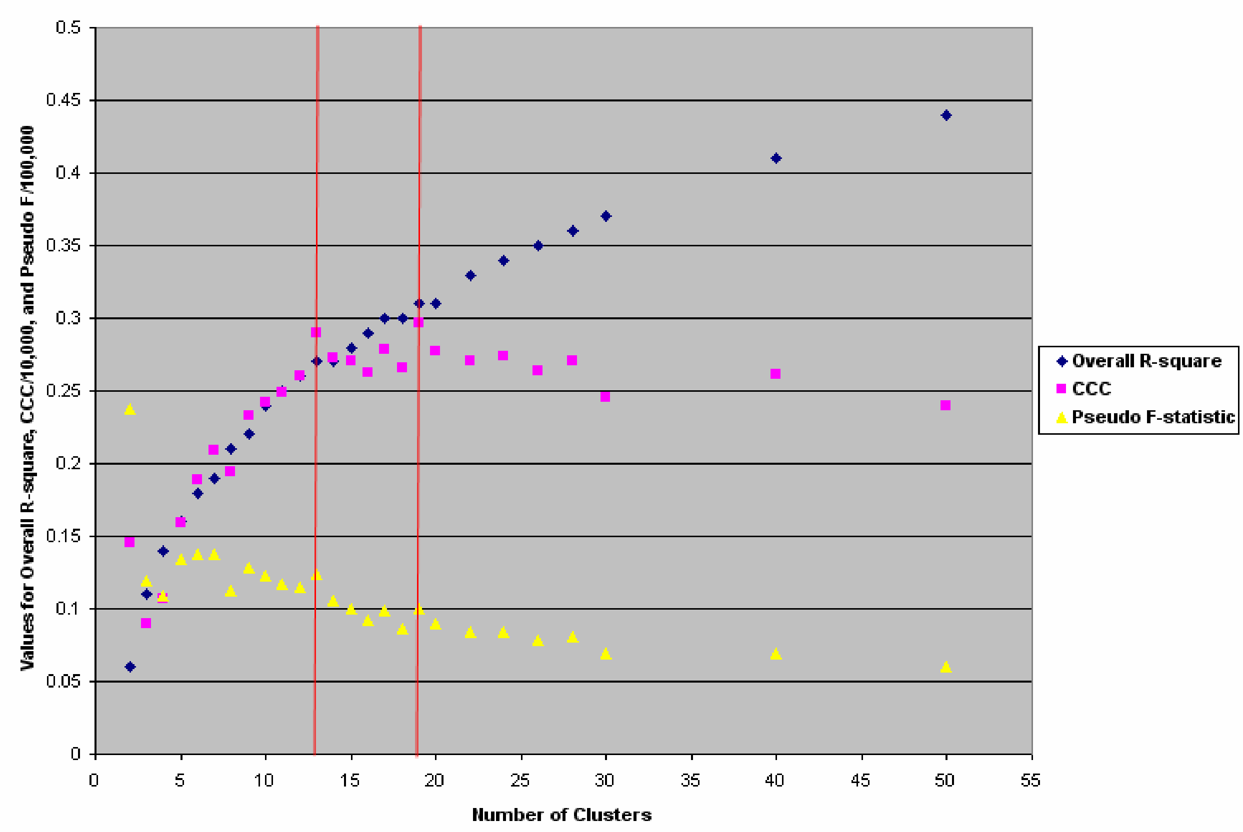


Figure 3.25. Plot of the overall r-square, cubic clustering criterion (CCC), and pseudo F-statistic values versus the number of clusters. This plot suggests that are somewhere between 13 and 18 distinct clusters (see red lines) within the dataset. Note: for presentation purposes, the CCC and Psuedo F-statistic values were divided by 10,000 and 100,000, respectively.

Clustering Results

Based on the diagnostic statistics and a visual examination of resulting classification units, we elected to use the groupings produced by the seventeen clusters (Figure 3.26). The initial seventeen groups for the CP and OZ were further stratified by accounting for spring and groundwater influences. AES polygons were given binary code that discriminated between those AES polygons with limited spring/groundwater influence and those with "significant" spring/groundwater influence. A "significant" spring/groundwater influence was based three criteria;

1. Contains a stream classified as coldwater
2. Contains a spring with a discharge greater than or equal to 10 cfs
3. Has a spring density greater than or equal to 1 spring per 10 mi^2

Any AES polygon that met one or more of these three criteria was given a binary code to denote a significant spring/groundwater influence (Figure 3.27).

Using the above criteria, an additional 12 groups were added to the initial 17 groups within the Central Plains and Ozarks. The resulting 29 groups for this part of the state, combined with the 10 delineated for the Mississippi Alluvial Basin, resulted in a total of 39 distinct AES-Types for Missouri (Figure 3.28). Maps and descriptions for each of the AES-Types can be found in Appendix 3.3.

Each AES-Type was assigned a unique identifier called the AES-Type Code. AES-Types are defined according to the input variable metrics and don't necessarily have to be in geographic proximity. AES-Types were assigned names based on the name of a major stream contained within the most representative or typical AES of a given AES-Type (see Figure 3.28 and Appendix 3.3 for Examples). Representative AESs were selected based on the distance from the cluster centroid. The individual AES that plotted closest to the cluster centroid, for each cluster, was selected as the most typical AES for a given Type. In cases where no stream name could be identified (this occurred only in the Mississippi Alluvial Basin) the name of a municipality or in one instance a major landform, contained in the representative unit, was assigned as the AES-Type name.

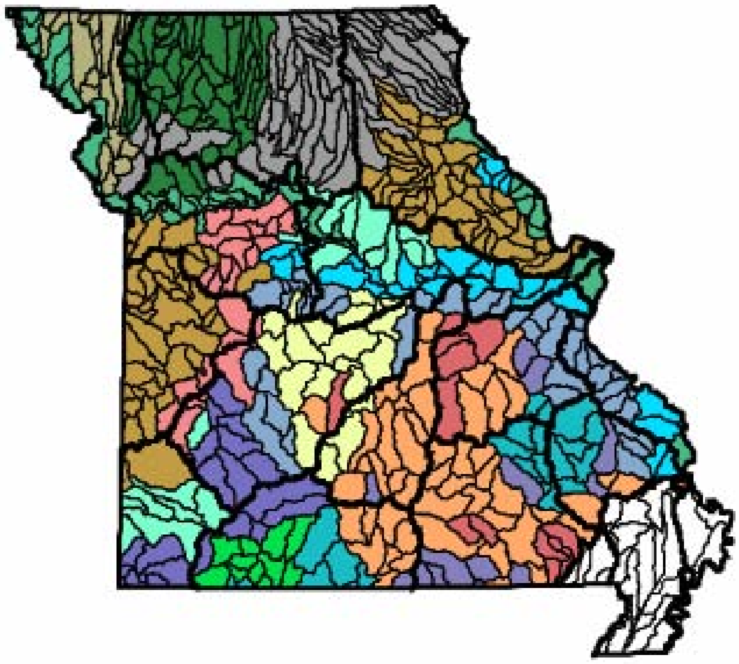


Figure 3.26. Map showing the spatial distribution of the initial 17 clusters identified within the Central Plains and Ozark Aquatic Subregions.

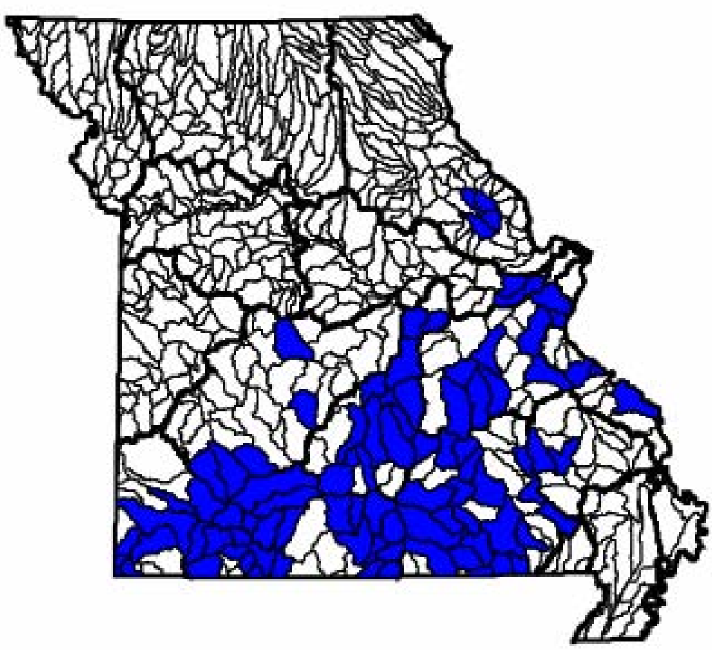


Figure 3.27. Map showing AES polygons (in blue) that were classified as having significant local spring and/or groundwater influences.

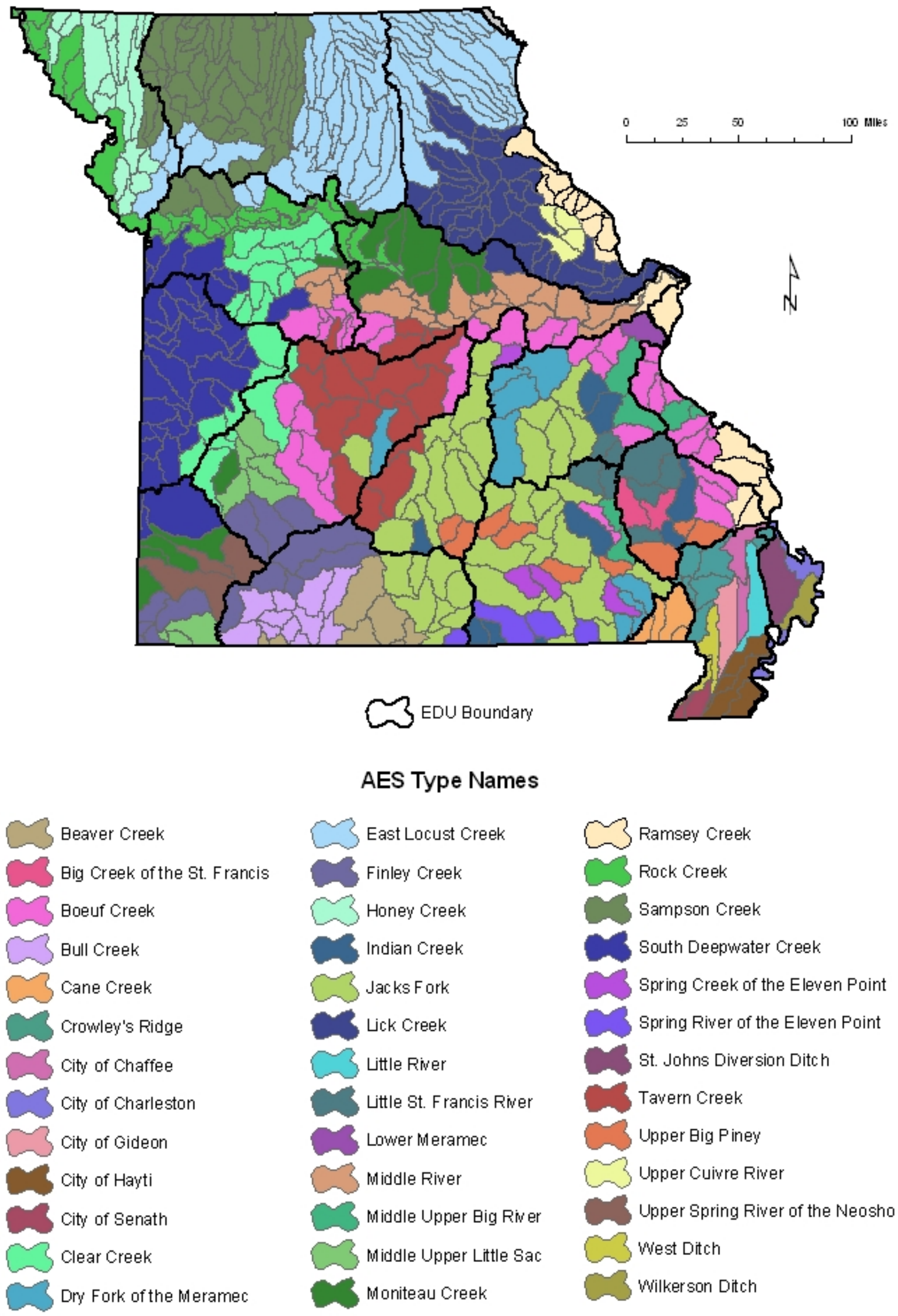


Figure 3.28. Map and names for all 39 Aquatic Ecological System Types (AES-Types) delineated for Missouri. Descriptions of each AES-Type can be found in Appendix 3.3.

3.7. Level 7: Valley Segment Types

Objective

Classify stream segments contained within the 1:100,000 National Hydrography Dataset into distinct valley segment types according to distinct combinations of factors known to individually and collectively influence local biophysical conditions.

General Description

Valley Segment Types (VSTs) are defined and mapped to account for longitudinal and other linear variation in ecosystem structure and function that is so prevalent in lotic environments. VSTs represent hydrogeomorphic units defined by local physical factors and their position in the stream network. They stratify stream networks into major functional components that define broad similarities in fluvial processes, sediment transport, riparian conditions, and thermal regimes. Each individual valley segment is a spatially distinct habitat, but valley segments of the same size, temperature, flow, gradient, etc. all fall under the same VST. Outside of the context of the upper levels of the classification hierarchy, we expect valley segment types to contain ecologically similar aquatic assemblages. However, within the context of the upper levels of the classification, we assume that individual valley segments, falling within the same VST, will contain aquatic assemblages that are similar in actual taxonomic composition.

General Methods

Stream segments within the 1:100,000 USGS/EPA National Hydrography Dataset (NHD) were attributed according to various categories of stream size, flow, gradient, temperature, and geology through which they flow, and also the position of the segment within the larger drainage network. These variables have been consistently shown to be associated with geographic variation in assemblage composition (Moyle and Cech 1988; Pflieger 1989, Osborne and Wiley 1992; Allan 1995; Seelbach et al. 1997; Matthews 1998). Each distinct combination of variable attributes represents a distinct VST. Stream size classes (i.e., headwater, creek, small river, large river, and great river) are based on those of Pflieger (1989), which were empirically derived with multivariate analyses and prevalence indices.

Mandatory Criteria
None

Software Used
ArcView 3.3
ArcInfo (workstation)
SAS 8.2
Microsoft Excel 2000

Baselayer, Source Data Used to Classify AES-Types (see Figure 3.29)

- National Hydrography Dataset (NHD) (1:100,000 USGS/EPA)
- Bedrock Geology of Missouri (1:500,000; Missouri DNR)
- Relief Grid (generated from a 30 meter DEM)
- Coldwater streams of Missouri (1:24,000 Missouri Department of Conservation)
- Losing streams of Missouri (1:100,000 USGS and Univ. of Missouri Geographic Resource Center)

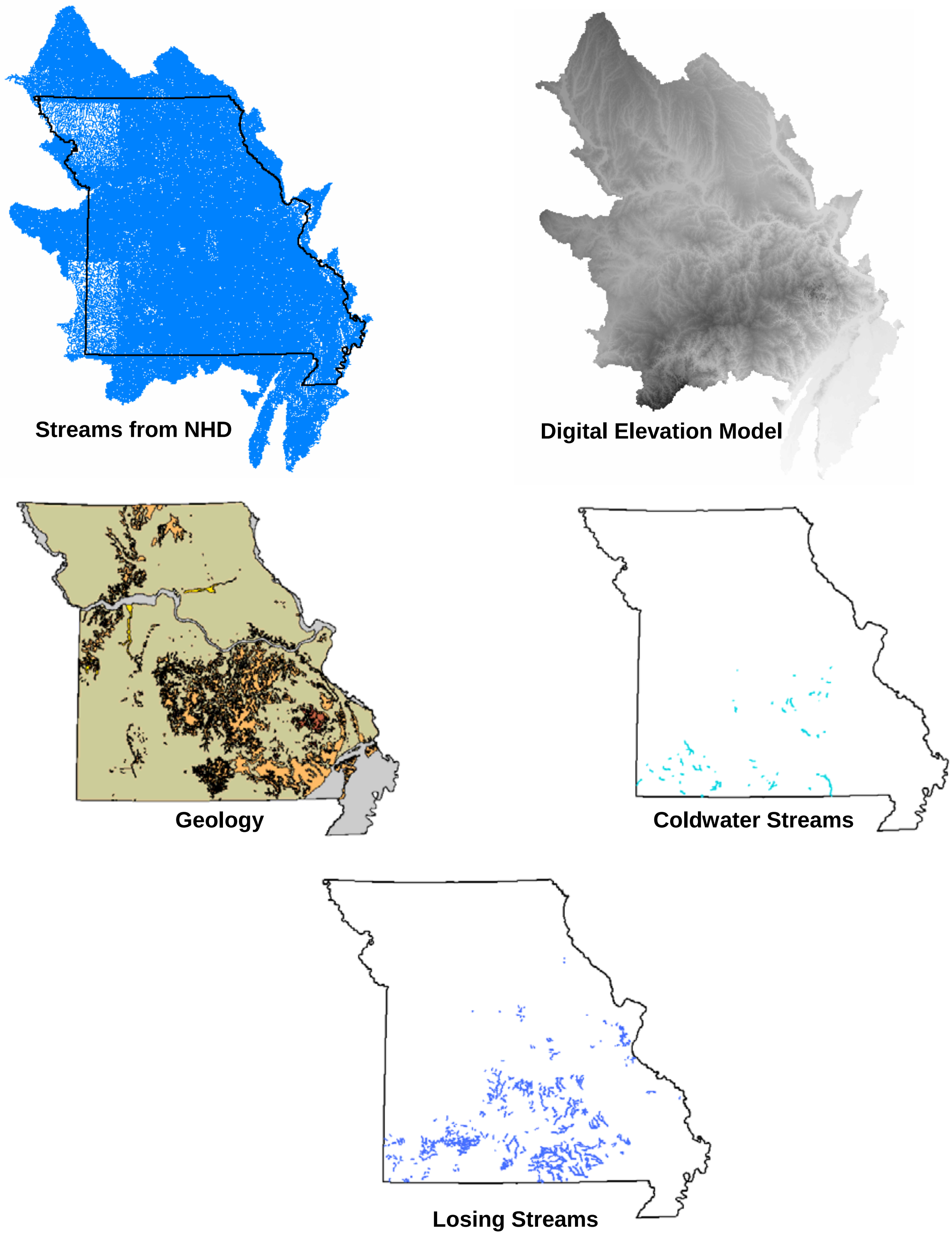


Figure 3.29. Maps showing the geospatial datasets used to classify the 1:100,000 NHD into distinct Valley Segment Types.

Detailed Methods

We began by using the 1:100,000 scale NHD as our base stream layer. At the initiation of our project only the *Initial Release* of the NHD was available. After acquiring the *Initial Release* of the NHD we first ran the Fixnhd.aml program that was developed by the Missouri Department of Conservation. This AML preprocessed each NHD file (within an individual 8-digit HU) by attaching a number of attributes from related tables to the arc attribute table (.aat). This facilitates the use of the NHD coverage by condensing the numerous related attribute tables within the NHD into single table containing a small subset of attributes required for processing. The AML also removes polygonal water body features resulting in a centerlined stream network.

Some areas in the 1:100,000 scale NHD were mapped with lower stream densities than most of the nation. These areas correspond to the boundaries of specific 1:100,000 scale topographic maps. These problem areas can be easily identified by viewing the stream networks across fairly large regions as evidenced by the rectangular nature of these low density area boundaries (Figure 3.30). These areas present problems when attempting generate standardized stream size classes because much of the contributing network is missing. We fixed these areas of lower stream density by generating the "missing streams" with a 30-m DEM (Figure 3.30).

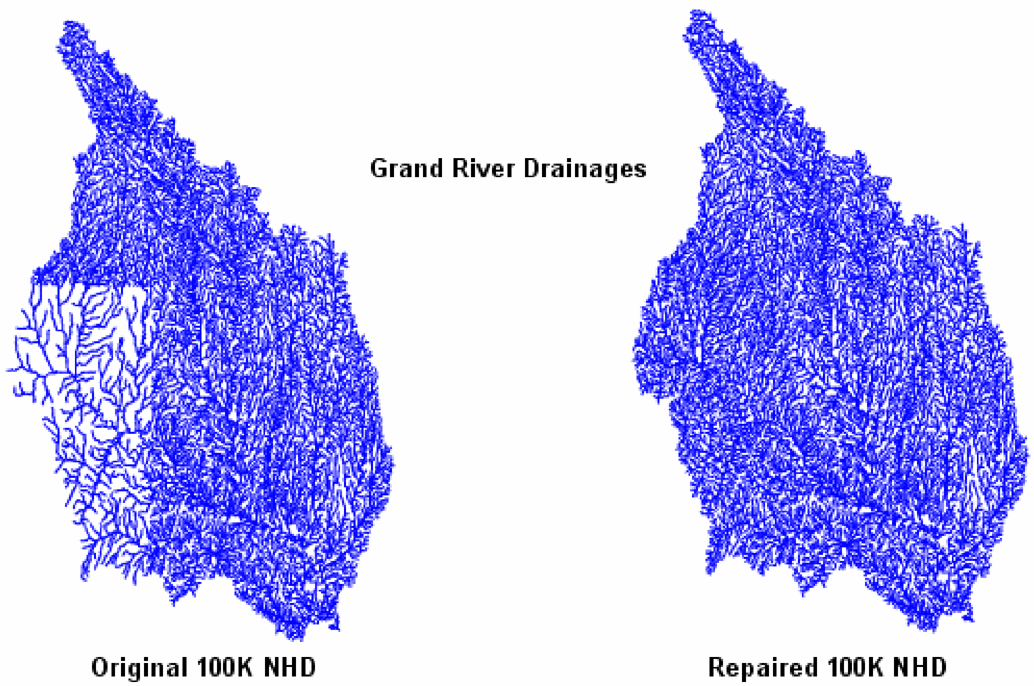


Figure 3.30. Example of a low-density area within the original National Hydrography Dataset (NHD) and the same area after repairing the networks using streams generated from a 30-meter DEM.

In addition to fixing areas of low stream density, we also occasionally fixed disconnected streams or entire stream networks when they were determined to be errors. This was accomplished by using digital representations of topographic maps or aerial photos as a cross reference. In instances where a clear connection could not be determined, we did not attempt to make a connection. Fixing these connections also helped improve the accuracy of stream size classifications.

Unique Identifier

To facilitate linking data to our Valley Segment Coverage we added a unique stream segment identifier called the Seg_ID. The Seg_ID was created by concatenating the 8-digit HU code with a unique value given to all segments contained within a given 8-digit HU. For ease of reading we placed a space between the 8-digit HU code prefix and the unique value portion of the Seg_ID. As a result, every stream segment within our Valley Segment Coverage has a unique Seg_ID, which functions in a similar fashion as a social security number or home address.

Coding Primary and Secondary Channels

To run stream ordering programs on the networks it was necessary to code and temporarily remove the secondary channels (loops and braids) from the primary channels (Figure 3.31). Coding primary and secondary channels can be a difficult task without doing extensive field verification. Several NHD table attributes that helped in determining the primary from the secondary channels were the flow attributes (permanent flow takes precedence over intermittent flow), stream name (a named channel takes precedence over an unnamed channel) and stream Level (the lowest level takes precedence). Another means of identifying the primary channel is to look at the angle created where two channels converge. The main path is generally the one that creates the least angle when looking upstream. Many instances arose where a judgment call had to be made. Areas presenting particular difficulty are those that have had their drainage patterns altered through channelization and ditching. It is often difficult to determine whether the majority of flow remains in the natural channel or has been diverted into a ditched portion; some ditches may even have a flow control gate. Without field verification places with multiple channels may not always be coded "correctly". These conditions should be recognized as a limitation in the data.

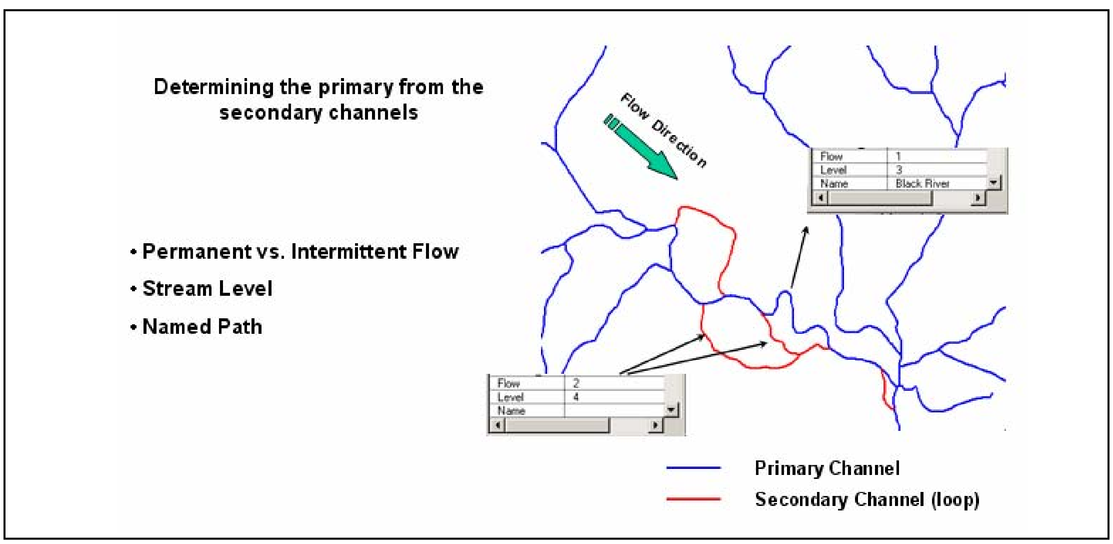


Figure 3.31. Example of mapping primary and secondary channels.

In addition to a single-line primary channel network, the stream ordering programs also required that pseudo-nodes be removed from the network. Once this initial preprocessing was complete, and a clean primary channel network was available, we were then able to commence with the stream ordering.

Stream Order

It has long been recognized that a wide array of structural features and functional processes, occurring within and along stream ecosystems, tend to change in a longitudinal continuum from the smallest headwaters to the largest rivers (Vannote et al. 1980). Consequently, studies designed to examine the potential influence of a given factor (other than drainage area) on the ecological character of streams, must somehow account for differences in stream size among sites.

Instead of using the more precise measures of drainage area or discharge most investigators have utilized discrete stream size classes (Sensu Horton 1945 and Strahler 1957) in order to more tractably account for longitudinal changes in the abiotic and biotic character of streams. The Strahler ordering system is certainly the most widely recognized and the one most often used by stream ecologists for research and management (Hansen 2001). However, Strahler order often underestimates stream size due to vagaries in drainage network structure (Hynes 1970). With the Strahler ordering system it is common to have lower order streams (e.g., 3rd) with substantially larger drainage areas than higher order streams (e.g., 5th). Recognizing this problem Shreve (1966) devised another measure of stream size, termed link magnitude, which overcomes this problem since it is much more precisely related to drainage area (Hansen 2001). Link magnitude simply reflects the number of first order stream channels within the watershed of a given stream segment.

Stream ordering consisted of running Arc Macro Language (AML) programs on the primary channel stream network to generate Strahler stream order, Shreve link magnitude, and downstream Shreve link magnitude. We used the Stream_o.aml program, developed by the US Forest Services Redwood Sciences Laboratory (Lamphear and Lewis 1994), to compute the Strahler Order for each arc in the network. We then used the Shreve.aml program, which was originally developed by the Missouri Department of Conservation and subsequently modified to work with this project, for computing Shreve link magnitude for each arc. This AML utilizes the Arcplot command TRACEACCUMULATE to accumulate the number of streams with a Strahler stream order of 1 above each segment.

Stream Size Class

The ordered networks were then classified into more general stream size classes. These size classes were based on Pflieger (1989), following his *Aquatic Community Classification System for Missouri*. Pflieger's size classes were based on fish community composition computed for over 1,600 sample locations. He defined his size categories using Strahler stream order. We modified this approach slightly and based our size class breaks on Pflieger's categories, but used Shreve link number when describing our intervals because link gives a more precise category than does Strahler order. Like Pflieger, our size classes were made relative to the surrounding Aquatic Subregion. Table 3.6 shows the stream size classes used for each of the Aquatic Subregions and Figure 3.32 shows a map of the resulting size classes for Missouri.

Table 3.6. Stream size classes used in the classification of Valley Segment Types.

Stream Size	Size Code	Central Plains (Shreve link range)	Ozarks (Shreve link range)	Missip. Alluv. Basin (Shreve link range)
Headwater	1	1-2	1-4	1-4
Creek	2	3-30	5-50	5-50
Small River	3	31-700	51-450	51-450
Large River	4	Greater than 700	Greater than 450	Greater than 450
Great River	5	Missouri and Mississippi Rivers	Missouri and Mississippi Rivers	Missouri and Mississippi Rivers

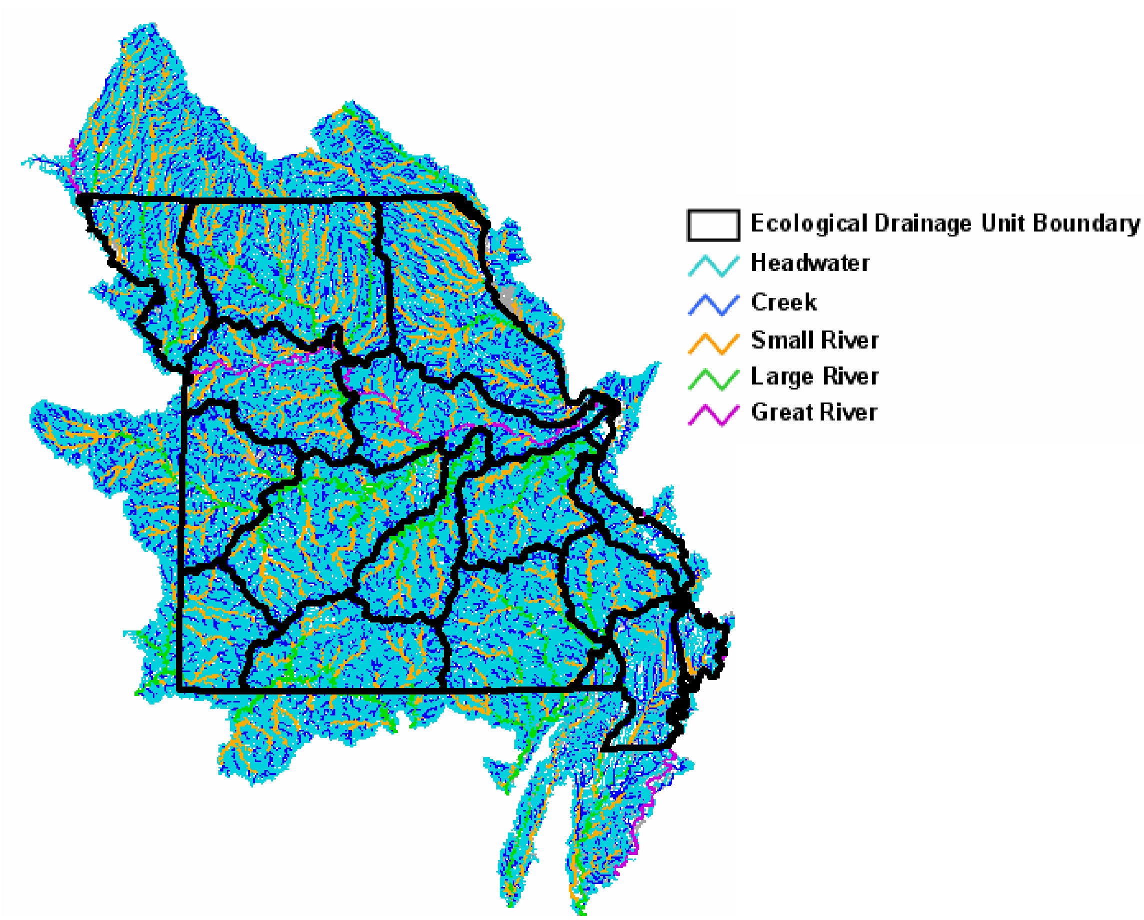


Figure 3.32. Map showing the five stream size classes used in the classification of Valley Segment Types for Missouri.

The size of the downstream confluence can have a significant influence on the aquatic assemblages of the influent stream (Osborne and Wiley 1992). To account for this phenomena, we also attributed stream segments within the NHD according to stream size discrepancy. This was accomplished by running the Dlink.aml program that was developed by the Missouri Department of Conservation and modified for use in this project. The Dlink program finds the Shreve link number of the next downstream segment for all segments in a network and attaches this value to each segment in a field called Dlink. We applied our stream size classes to the Dlink field to create a new field for downstream stream size called Dsize. We created distinct attribute categories for the different available combinations (i.e. headwater connecting to a creek or a headwater connecting to a small river, etc) (Table 3.7). These eleven classes (0-10) were also condensed into just two classes, in order to simply identify stream segments that flow into a segment falling into a larger size class.

Table 3.7. Size discrepancy classes.

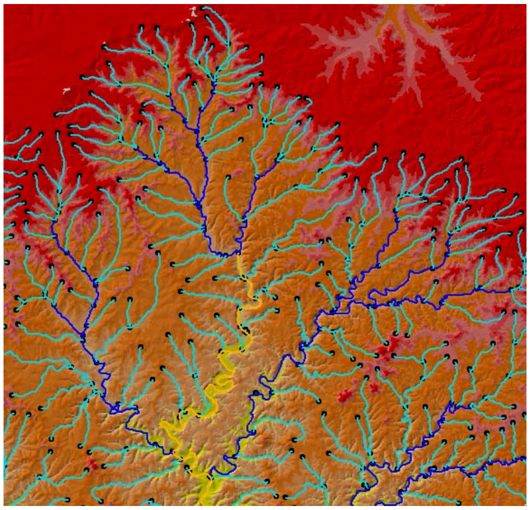
Size Discrepancy	Size Discrepancy Code (11 Class)	Size Discrepancy Code (2 Class)
None	0	0
Headwater – Creek	1	0
Headwater – Sm. River	2	1
Headwater – Lg. River	3	1
Headwater – Great River	4	1
Creek – Sm. River	5	1
Creek – Lg. River	6	1
Creek – Great River	7	1
Sm. River – Lg. River	8	0
Sm. River – Great River	9	1
Lg. River – Great River	10	0
Disconnected streams	-1	-1

<u>Stream Gradient and Relative Gradient</u>

Stream gradient has long been recognized as a principle adjustable property of rivers that is often found to be associated with numerous abiotic and biotic factors within streams (Hack 1957; Knighton 1998; Nino 2002). To calculate gradients for each stream segment, we first preprocessed a 30-m DEM in order to fill all sinks within the DEM. Sinks represent single or multiple grid cells within a DEM that cannot be assigned a flow direction. To fill all sinks we used the Fill Sinks algorithm that is part of the Hydrologic Modeling tool set in ArcView. The resulting depressionless DEM was subsequently used to calculate gradients for each arc within the modified 1:100,000 NHD.

Stream gradient was calculated and applied in two different ways. The first and most straightforward method involved calculating gradient for every confluence-to-confluence segment and is represented in meters per kilometer (Figure 3.33). This was accomplished by draping the stream network over a 30-meter resolution digital elevation model (DEM) and getting the elevation for the node of each stream confluence. Once the upstream and downstream elevations for every stream segment were acquired, the difference between them was divided by the segment length (in meters) and multiplied by 1000 to get a gradient in meters per kilometer.

Method 1



Method 2

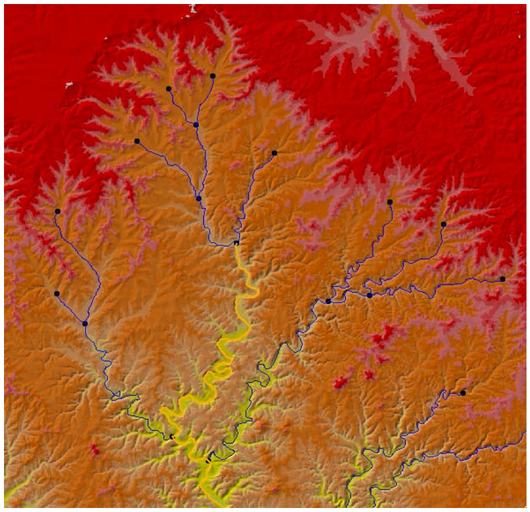


Figure 3.33. The map on the left shows all streams and confluence nodes on top of a DEM. The map on the right shows the same area with the headwaters removed and the subsequent pseudo nodes also removed. The maps illustrate the different segments for which gradients were generated for streams classified as Creek or larger.

In cases where a major reservoir had inundated a stream, and the DEM represented the water surface elevation, we interpolated gradients through the inundated stream network (Figure 3.34). We did this to get an approximation of the natural system that existed before inundation. In this process the largest streams are interpolated first followed successively by the next largest and so on until all segments have interpolated elevation values. Once complete, the confluence elevations are used to determine gradient on each segment as described above.

There were instances where the 1:100,000 stream network and the DEM did not correspond perfectly. In some of these instances a portion of the stream fell on valley wall instead of the stream bed, as depicted in the DEM, which produced an elevation on the downstream node that was higher than the usptream node. This resulted in a negative stream gradient. When this occurred we looked at the stream segment in question and the DEM and manually corrected the erroneous elevation and recomputed gradient. In certain instances an appropriate elevation correction could not be determined manually. In these cases we calculated the gradient equal to zero and flagged these segments. These problems were most prevalent in relatively low relief areas.

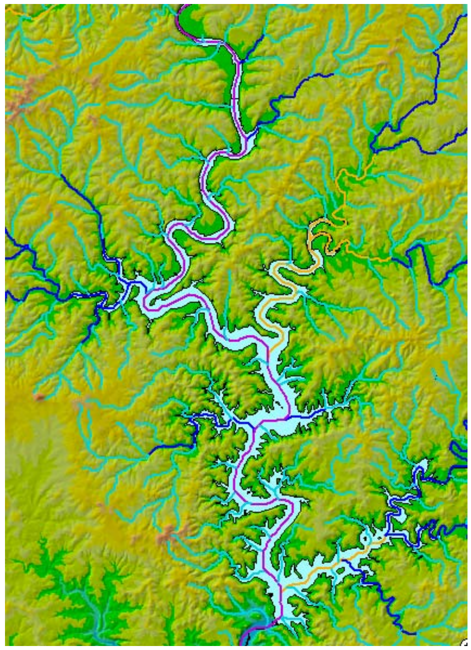


Figure 3.34. Example of an area in which confluence elevations were interpolated through major reservoirs because the DEM elevations represented the elevations of the water surface.

The second method of calculating stream gradient involved computing gradient over longer distances for streams that were classified as Creek or larger. This was necessary because the drop in elevation between two closely spaced confluences on these larger streams was often less than 1 m, which is the vertical precision of the DEM. By computing gradient over a longer distance the drop is more likely to be at least a meter allowing a gradient other than zero to be to be determined. To accomplish this we removed all headwater segments from the stream network and then removed all of the psuedonodes (see Figure 3.33). Gradients were then obtained for these much longer segments and these gradients were then applied back to all of the the arcs that had initially made up that segment prior to removing headwater streams. The raw stream gradients of this second method were also placed into relative-gradient categories of low, medium or high (Figure 3.35). These gradients are relative to both stream size and Aquatic Subregion.

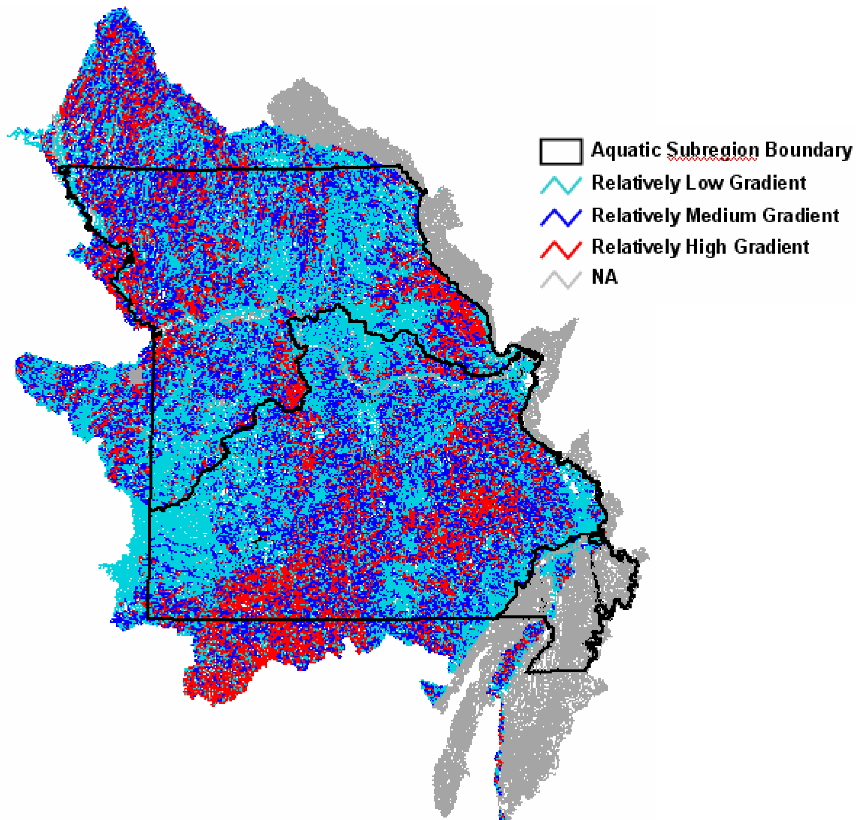


Figure 3.35. Relative stream gradients for Missouri. Gradients are relative to both stream size and Aquatic Subregion.

Stream Flow

Stream flow (intermittent versus perennial) was an attribute that was already contained within the 1:100,000 NHD. For our purposes, this attribute was translated into a binary code. In areas of low stream density where we “repaired” the NHD (described earlier) the resulting stream reaches we added were assigned an estimated flow. To accomplish this we found the “average” Shreve link number at which flow transitioned from intermittent to perennial within the same Ecological Drainage Unit. This Shreve link value was used to assign flow to all added reaches. Segments having Shreve link values smaller than this “average transition to perennial flow” were assigned a code for intermittent flow and all link values equal to or larger were assigned a code for permanent flow.

Inundated stream segments in the NHD are typically coded as having permanent flow. We coded streams that are now inundated reservoirs according to the flow they would likely have if inundation had not occurred. This was accomplished by looking immediately upstream of the inundated segments and applying the upstream flow codes to the inundated segments.

Losing Attribute
A “losing stream” is a stream that loses some or all of its surface flow to the underlying groundwater system. Stream segments were coded as to whether or not they are losing segments. The losing attribute was taken from the 1:100,000 state hydrography coverage of Missouri, which was originally produced by the USGS in 1995. The Losing attribute was added to this coverage by James Harlan (Geographic Resource Center, University of Missouri) in 1997.

Temperature
A temperature code was assigned to every stream segment based on a coverage of known coldwater stream segments developed by the Missouri Department of Conservation. Stream segments were coded as being either “cold” or “warm”. Specific temperature ranges were not available.

Geology
Bedrock geologic type codes were assigned to each stream segment by assigning the general geologic type that the majority of the segment is flowing through (Figure 3.36). This approach is used to avoid having to break a stream segment into numerous small segments every time it crosses a geologic boundary. A good example of a stream segment that would otherwise have to be broken is a segment that flows along a geologic boundary and frequently crosses back and forth from one geologic type to another. Segments with any igneous component were identified as such in a separate field.

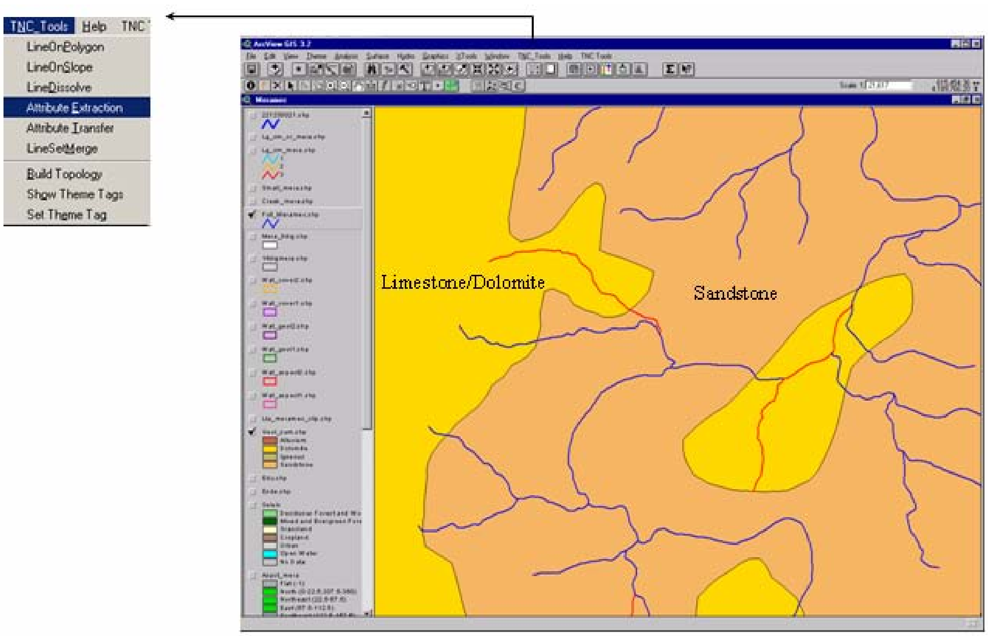


Figure 3.36. Example of how the general geologic type was assigned to stream segments based on what the majority of a stream segment was flowing through. In this view the segments in red have been coded as Limestone/Dolomite, whereas all others are coded as Sandstone.

Valley Wall Interaction as a Surrogate for Potential Bluff Pool Habitat

Limestone bluffs as high as 150 feet border many of larger streams both within and along the periphery of the Ozark Aquatic Subregion. In many places, the pools adjacent to these bluffs (i.e., bluff pools) are often extremely deep and contain large complexes of boulders. These bluff pools have been identified as important flow refugia and overwintering habitat for many species and are also a key habitat for the spectaclecase mussel (*Cumberlandia monodonta*) (Peterson 1996; Baird 2000). Locations where the stream channel comes in contact with the adjacent valley wall and locations where the stream pulls away from the valley wall are typically where bluff pools occur (Robb Jacobsen, personal communication). As a surrogate for directly mapping bluff pools, we mapped point locations where the stream came in contact with, and moved away from, the adjacent valley wall. These valley-wall interaction points were only mapped along Small and Large Rivers within and also along the periphery of the Ozarks. We then counted the number of valley wall interaction points within a 2.5 Km search radius from the centroid of each stream segment (Figure 3.37). These valley wall interaction counts were then placed into relative categories of low, medium or high based on the standard deviation of the values within the surrounding EDU.

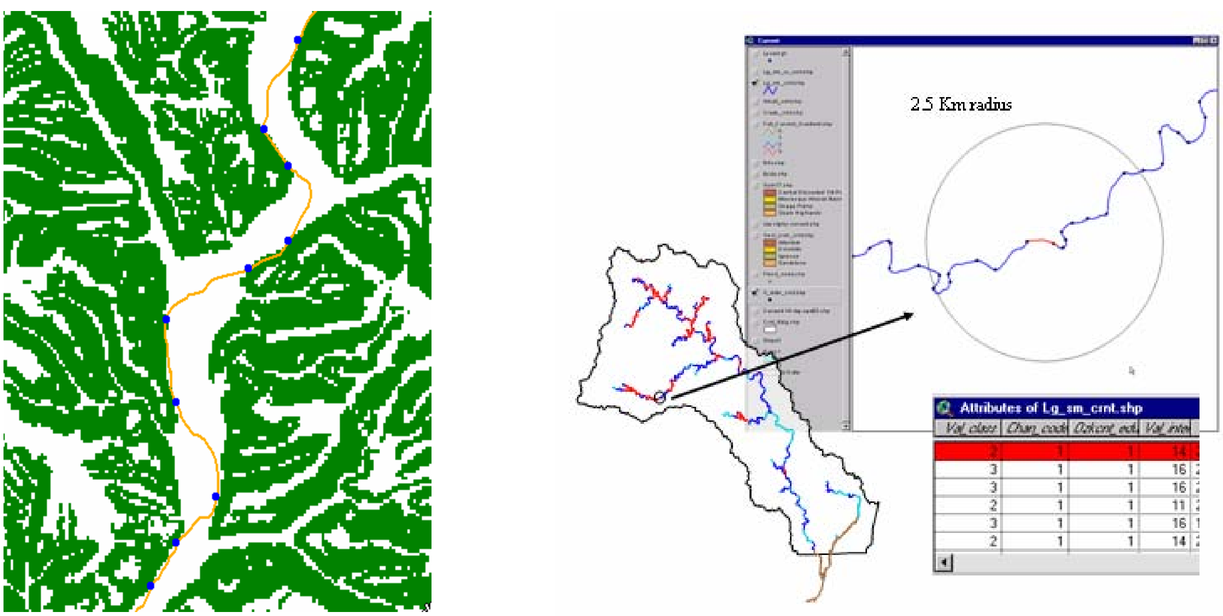


Figure 3.37. The map on the left shows a slope grid (white areas have less than a 5% slope and green areas have greater than a 5% slope) and the point locations where streams hit and pull away from the valley wall. These point locations are mapped and used as a surrogate for potential bluff pool habitat. The map on the right shows how the number of valley wall interaction points are counted within a 2.5 kilometer search radius from the centroid of each segment.

Floodplain Segment

Stream segments that begin in uplands and then flow across the floodplain of a larger stream often exhibit distinctly different physical characteristics, and thus habitats, than the upland portions. As a result, these "floodplain" segments were given a distinct code so that they could be easily identified. Headwater and Creek segments were coded as floodplain segments if 250 meters or more of their length flowed across the floodplain of a stream classified as either Small, Large, or Great River (Figure 3.38).

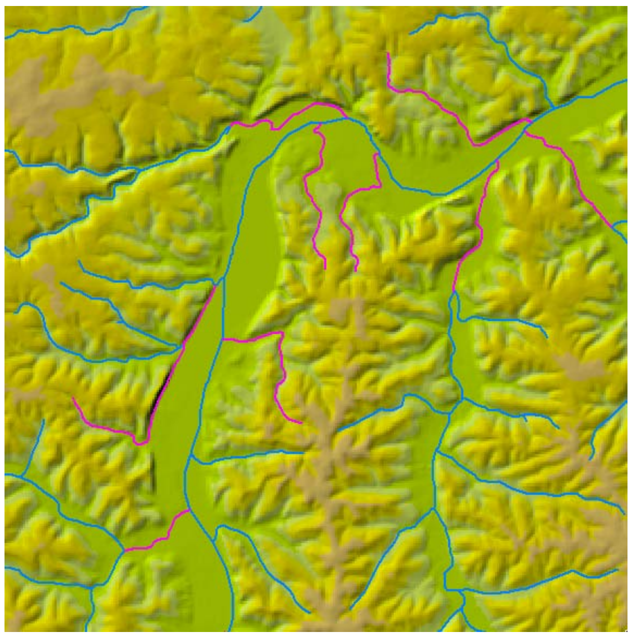


Figure 3.38. Map showing an example of floodplain segments (in magenta), which are defined as headwater or creeks that have 250 meters or more of their length cutting across the floodplain of a stream classified as either Small, Large, or Great River.

Joining attributes back to the full network

Once the primary channel network was completely classified, we joined the attributes back to the full stream network, which includes all primary as well as secondary channels (Figure 3.39). Finally, the individual codes for each of the attributes were concatenated to create the Valley Segment Type (VST) code. Each distinct combination of individual attribute codes represents a distinct VST (Figure 3.40). The boundaries between different VSTs can be determined by a single attribute (e.g., change in stream size category) or a combination of attributes (e.g., change in geology and gradient). Different combinations or subsets of variables can be used to create different VSTs to meet a variety of research and management needs. Each individual valley segment is a spatially distinct habitat, however, all valley segments of the same size, temperature, flow, gradient, etc... fall under the same Valley Segment Type (Figure 3.41).

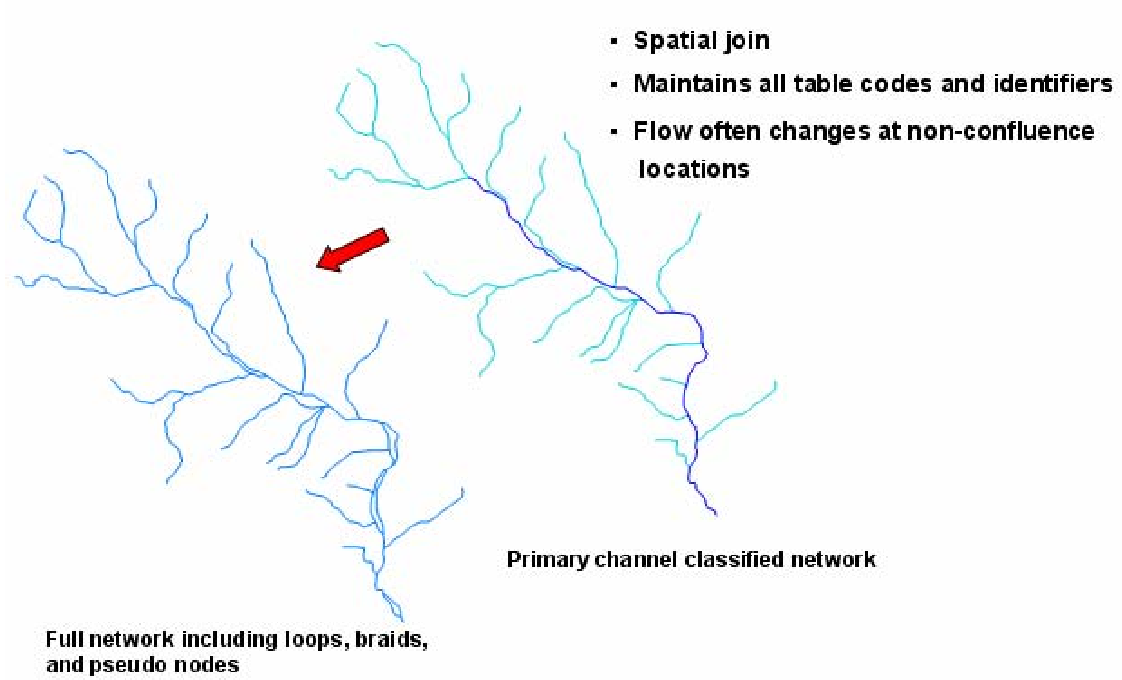


Figure 3.39. An example of how the classified primary channel network was joined back to the full network, which contained all of the secondary channels.

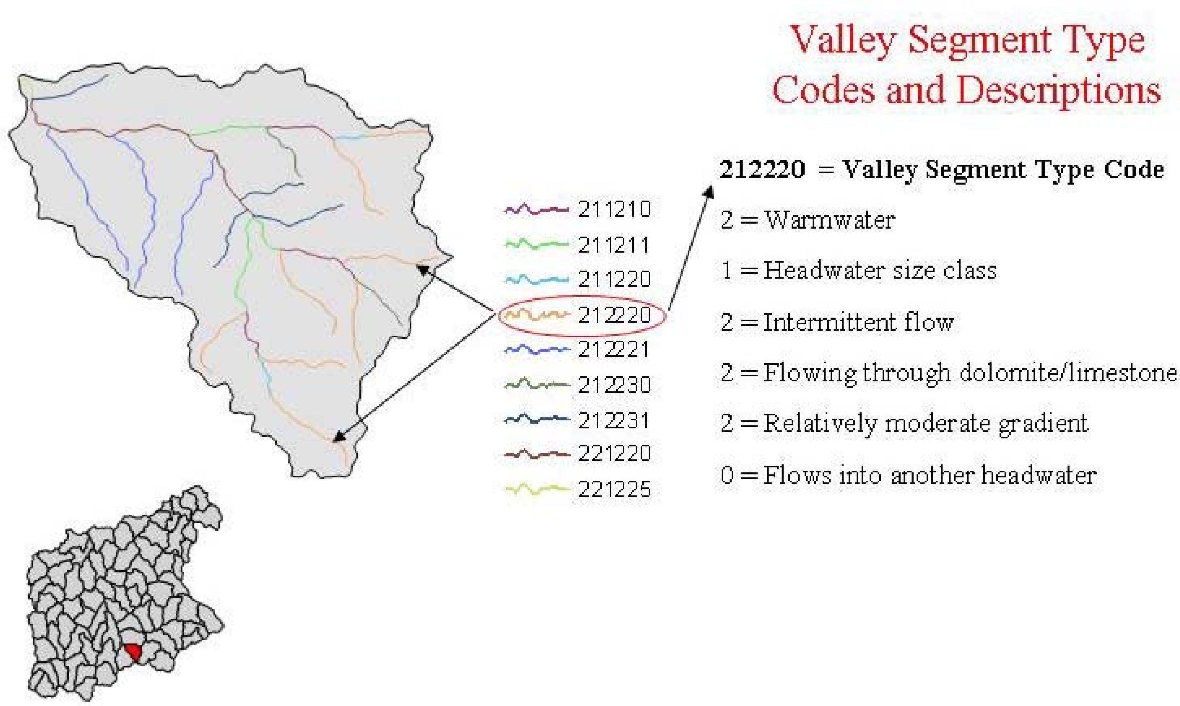


Figure 3.40. An example of Valley Segment Types (VSTs) for a single 12-digit hydrologic unit. The placement and value of each number in the VST code has meaning and can be deciphered to make informed decisions on the spatial arrangement and relative abundance of stream types across any geographic area of interest.

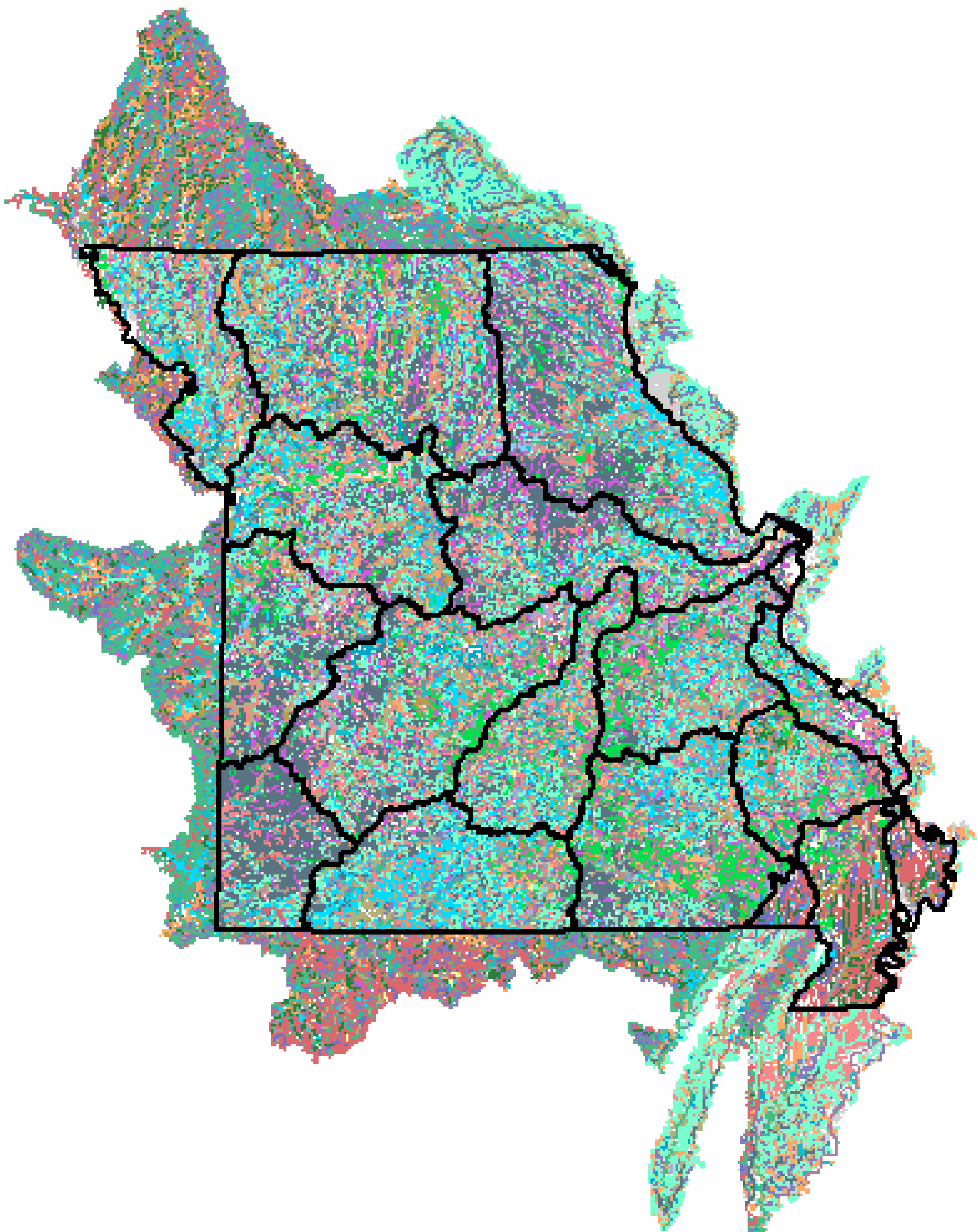


Figure 3.41. Map showing streams classified into distinct stream Valley Segment Types for Missouri.

3.8 Level 8: Habitat Types

Ecological units within the final level of the hierarchy, Habitat Types (e.g., high-gradient riffle, lateral scour pool), are simply too small and temporally dynamic to map within a GIS across broad regions or at a scale of 1:100,000. However, we believe it is important to recognize this level of the hierarchy since it is a widely recognized component of natural variation in riverine assemblages (Bisson et al. 1982; Frissell et al. 1986; Peterson 1996; Peterson and Rabeni 2001).

3.9 Discussion and Limitations

Without question, the first step to effective resource management is having an accurate inventory of the resources you intend to manage and the only way you can generate an inventory is to have a classification system (Fajen 1981; Lotspeich and Platts 1982). We fully recognize that by classifying the natural world into discrete units we are often placing somewhat arbitrary boundaries on a continuum of change (Whittaker 1962; Grossman et al. 1998). However, you cannot generate an inventory for a continuum since every value is unique. We agree with Orians (1993) and Angermeier and Schlosser (1995) that if we are going to be effective in our efforts to conserve biodiversity we are going to have to demonstrate the extent of the problem and thus the need for new conservation policies and actions. The only way of demonstrating such a need is through a systematic accounting of the various elements of biodiversity, not simply species, but also the ecosystems and habitats that sustain these species. As Angemeier and Schlosser (1995) point out, only then will we be able to answer fundamental questions like; How many types of ecosystems/assemblages exist? How many of each type remain? Where are they? Which ones are most imperiled? Failure to answer these questions will relegate the conservation of biodiversity to haphazard preservation of fragments of disintegrating systems (Angermeier and Schlosser 1995).

Since we cannot directly map biodiversity, we must identify suitable surrogates for assessing conservation gaps. Ideally, we should use both biotic and abiotic targets. Abiotic targets should be based on classification systems that define distinct ecosystem/ecological units. When defining these units we must account for structural, functional, and compositional variation across the riverscape (Noss 1990), and also ensure that at each level of the hierarchy we are delineating interacting systems in order to meet one of the fundamental components in any definition of an ecosystem (Grumbine 1994). The difficult part is doing the necessary detective work to identify those landscape, watershed and local factors responsible for natural variation at numerous spatial and temporal scales. The fact that evolutionary history plays such a dominant role in determining geographic variation in community composition dictates the need for a separate classification framework for terrestrial and freshwater ecosystems (Matthews 1998).

We went to great lengths in our efforts to incorporate existing ecological theory and objective statistical approaches into our classification framework in order to ensure that we were able to account for all three forms of distinctiveness (structure, function, and composition) at multiple spatial scales. However, there is room for improvement if we can overcome some important data limitations. More detailed geology and soil data would allow us to more accurately characterize both watershed and local conditions. Unfortunately, high-resolution geologic data is not standardized among states, which causes problems for creating a seamless classification across state boundaries. Also, the higher resolution 1:24,000 SSURGO soil data have not been converted into a GIS format for many counties across the nation, requiring the use of the 1:250,000 STATSGO soils data.

Stream temperature is likely one of the most influential ecologial parameters influencing the biological composition of streams and is strongly influenced by a wide variety of anthropogenic factors (Ferguson 1958; Huet 1959; Magnuson et al. 1979; Reynolds and Caterlin 1979). At present, the thermal regime of most of Missouri's streams (especially in the karst geology of the Ozarks) can only be depicted as either cold or warm. New technologies, such as Forward Looking Infrared Radar imagery (FLIR) provide a powerful tool for more precisely characterizing thermal regimes of surface waters. A project in Oregon has revealed that FLIR data can be used to remotely map stream temperatures to within 1 °C for large regions (Torgersen et al. 2001). Using this technology during mid July to early August we could generate a surface temperature datalayer for Missouri that would allow us to more precisely classify Missouri's streams into maximum summer thermal categories (e.g., headwater, maximum summer temperature: 17-19, 20-22, 23-25, 26-28, 29-31, >31). We firmly believe that a statewide stream temperature datalayer would advance our understanding and conservation of Missouri's stream resources more than any other datalayer.

Finally, we also need to take steps to link flow, physical habitat and water chemistry data to the NHD. Having spatially explicit data for these critical ecological factors would allow us to more precisely identify significant associations between landscape features and instream habitat. The problem with completing such a task is either the complete lack of data or the lack of data standards. Long term hydrologic data from USGS gaging stations is mainly available for larger streams and the density of the gage network is insufficient for characterizing more subtle differences in hydrologic regimes related to more subtle differences in watershed conditions. Physical habitat and water chemistry data have been collected by a wide variety of state and federal agencies and academic institutions over the years and the lack of a standardized schema for collecting and reporting these data is a major impediment to merging data from these various sources into a single statewide or nationwide geospatial dataset. Nonetheless, efforts must be taken to link existing sampling data to nationally standardized geospatial datasets like the NHD and at the same time national standards for collecting, storing, and reporting these data must become a priority if we are ever going to make progress in sharing this critical environmental data.

Finally, our classification units should not be blindly accepted. Efforts must be taken to empirically validate the classification hierarchy, especially the Aquatic Ecological System and Valley Segment Types. This will require the spatially extensive and temporally intensive physicochemical and biological data collected at reference-quality sites. Fortunately, some of these very data have begun to be collected across Missouri by the Missouri Department's of Conservation and Natural Resources, as part of their joint Resource Assessment and Monitoring program. As these data become available and are appropriately analyzed, we will take efforts to improve and modify our classification, as necessary.

CHAPTER 4

Predicting Species Distributions

Understanding and predicting the composition of local biologicial communities across the landscape is one of the main challenges confronting ecologists, including stream ecologists. – N. LeRoy Poff

4.1 Purpose

- Only 0.03% of the stream miles in Missouri have been sampled, and much of this data is spatially and temporally biased. Predicted distribution maps provide spatially comprehensive biological data at the finest level of our gap analysis (individual stream segment), which is a resolution that managers can comprehend and at which conservation action typically takes place.

- Since we cannot directly measure or map biodiversity, species within those taxa for which adequate sampling data is available, and the associated assemblages, must serve as surrogate biotic targets for biodiversity conservation, which complement the abiotic targets.

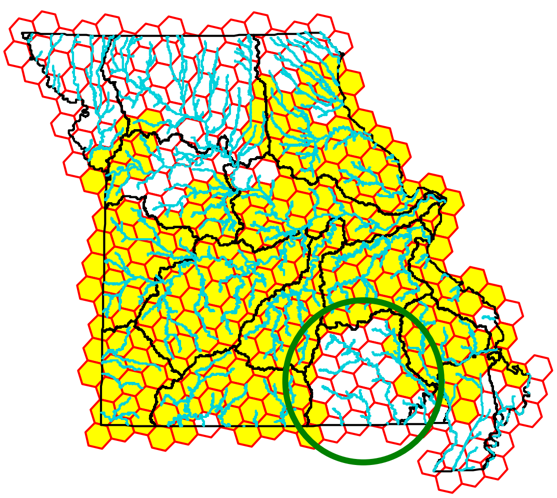
- Conservation values of society are largely biologically based. The public, legislators, and even scientists can more readily comprehend and relate to biologically-based assessments than other measures of biodiversity (e.g., habitat or processes).

4.2 Introduction

Gap analysis is a conservation assessment methodology that compares the distribution of several elements of biological diversity with areas managed primarily for native species and natural ecosystems (Scott et al. 1993). To accomplish this task, it is necessary that GAP develop detailed and relatively high-confidence distribution maps of individual animal species for comparison with maps of land stewardship and management status. These comparisons are used to assess the habitat area and relative percentage of the distribution of each species with the different categories of stewardship and biodiversity management status (Csuti and Crist 1998).

There are three types of distribution expressions: 1) *actual distribution,* which is based on exhaustive, long–term surveys that are very rare; 2) *known distribution*, which is based on current knowledge of where the species has been found and is usually incomplete, and 3) *predicted distribution*, which combines known distribution and knowledge of habitat associations of the species to extrapolate to unsampled areas where the species is expected to occur (Csuti and Crist 1998). It is simply impractical to map the distribution of hundreds of species through intensive field surveys across entire states, regions, or nations (Scott et al. 1996). GAP therefore makes use of existing information on range limits and wildlife habitat relations to develop spatial statements of the probability of a species being present within a given mapping unit that represent appropriate habitat as understood from current knowledge of the species and the ability to map its habitat. (Csuti and Scott 1991, Scott et al. 1991, 1993, Butterfield et al. 1994). The underlying assumption of GAP's predicted species distribution maps is that a species has a relatively high probability of occurring in appropriate habitat types that are within its known or predicted geographic range (Csuti and Crist 1998).

Existing GAP standards (See Csuti and Crist 1998) for delineating the geographic ranges and predicting species distributions within their range are not suited for riverine biota. Geographic ranges for terrestrial species are delineated using standardized grids like the Environmental Protection Agency's (EPA) Environmental Monitoring and Assessment Program (EMAP) hexagons, which is a regular equal-area 635 km2 hexagonal grid, developed specifically for EMAP (White et al. 1992). Since these grid-based methodologies do not account for, and consistently cross over, watershed boundaries they can be especially problematic for obligate aquatic species that often have very discrete and disjunct geographic ranges that correspond to drainage systems or watersheds (Pflieger 1997; Master et al.1998; Matthews 1998) (Figure 4.1).



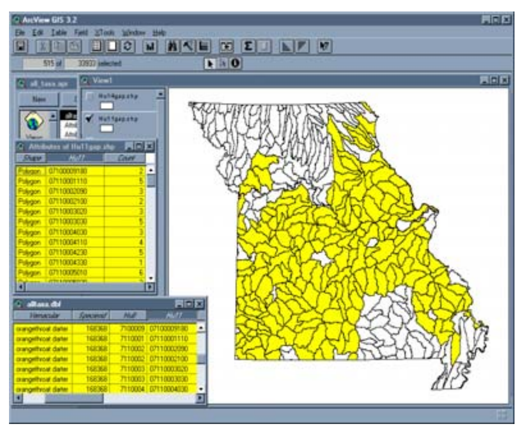


Figure 4.1. Maps depicting the geographic range of the orangethroat darter (*Etheostoma spectabile*). Map on the left was created by intersecting collection records with the EPA EMAP hexagons. The circled area illustrates why regular grids are not suited to mapping ranges of obligate freshwater species since they cross drainage divides. In this instance, the range map places the orangethroat darter in the Current, Black, and Eleven Point drainages, despite the fact this species does not occur within these drainages. The map on the right shows a more appropriate geographic range for the orangethroat darter that was generated by intersecting collection records with the USGS/NRCS 10-digit hydrologic units.

The necessity to account for watershed boundaries when mapping geographic ranges of riverine biota can best be illustrated with the following example. The Huzzah Creek and West Fork of the Black River watersheds are two watersheds that share a common drainage divide within the Ozark Plateau physiographic region of Missouri (Figure 4.2). Despite sharing a common drainage divide the outlets of these two watersheds are separated by over a thousand miles of stream, much of this being the Mississippi River. Both watersheds have been intensively sampled for fish, crayfish, mussels and snails. Comparing species lists of these two watersheds reveals the dramatic influence that millions of years of isolation can have on generating differences in the species composition among watersheds. A total of ninety-seven fish, crayfish, mussel and snail species occur within the Huzzah Creek watershed while only forty-two species occur within the West Fork of the Black River watershed. What is most striking is that only twenty-nine of these species are found in both watersheds. Mapping geographic ranges with a regular grid, like is done in the terrestrial component of GAP, significantly obscures differences in the species composition of these two watersheds. Such dramatic differences in composition, which consistently occur among adjacent watersheds, require the geographic ranges for riverine biota be mapped using watershed polygons or hydrologic units and actual collection data, not by visually inferring geographic ranges from field guides or taxonomic references.

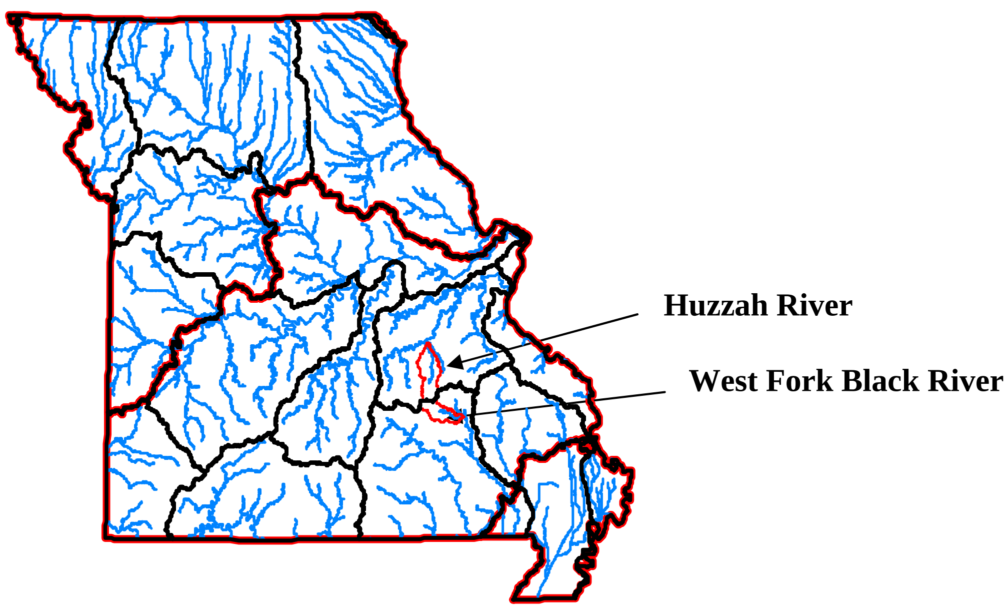


Figure 4.2. Map showing the locations of the Huzzah and West Fork Black River watersheds. See above text for a discussion of the differences in species composition among these two watersheds.

To generate predictive distribution maps for terrestrial biota GAP selects grid cells containing suitable land cover types (typically based on 30-meter resolution Thematic Mapper Satellite data) within the geographic range of each species. These models often include additional predictor variables or constraints based on other physical or spatial data, such as elevation, aspect, soils, or hydrography, for which existing geospatial data are available (Csuti and Crist 1998). Again, this grid-based approach is not ideally suited for predicting the distribution of riverine biota where vector-based hydrography coverages provide a more realistic depiction of the linear structure of natural stream networks and the upstream or downstream distributional limits of riverine species (Dunham et al. 2002). Furthermore, while the distribution of riverine biota are certainly influenced by those factors commonly used to predict the distributions of terrestrial biota (e.g., land cover, soils, and elevation), more often their distributions are more closely associated with other environmental factors like stream size, gradient, temperature, and flow and therefore require additional geospatial datasets and predictor variables (Moyle and Cech 1988).

A third major distinction between the predictive distributions produced in our project and those produced in the terrestrial component of GAP pertains to what the final maps actually portray. Typically, predictive distribution maps generated for terrestrial biota reflect the present-day distribution of a species (however see Oregon Gap Analysis Project; Kagan et al. 1999), whereas our predictions reflect a combination of both historic and present distributions. This difference in end products is the result of several confounding factors that inhibit our ability to strictly map present-day distributions for riverine biota. First, the lack of sufficient collection records inhibits our ability to precisely document the present geographic range of a species. Attempts to use only “recent” collections (e.g., collections after 1970) resulted in grossly restricted ranges for many species due to the omission of a high percentage (approximately 50%) of the already limited number of collection records. Certainly, in many respects it would be desirable to only map the present geographic range of each species since this range represents existing opportunities for proactive conservation. This issue is most critical for species exhibiting range contractions or shifts since assessment statistics (e.g., richness or number of species of special concern) within watersheds or stream segments where species have been locally extirpated will be somewhat inflated. However, accurately documenting range contractions or shifts is difficult at best and requires an immense number of samples. Even in those instances where there appeared to be sufficient evidence of a range contraction we found that a closer examination of the data revealed that many stream segments, which could potentially support populations within the watersheds where a species was suspected of being extirpated, had actually never been sampled. Consequently, we could never definitively state that a certain species no longer existed within a given watershed. We determined it would be logistically impossible to scrutinize the collection records of every species that is suspected of experiencing a range contraction or shift in order to strictly map the present range of each species. After consultation with taxonomic experts we decided that such an exercise would likely introduce more undesired omission errors into our range maps and our predictive distribution models, and ultimately that it is more informative, at least initially, to work within the constraints of the data to map overall potential distributions.

Recent collection records by the Missouri Department of Conservation (MDC), not included in our sampling database, further support our decision of using all collection records to map the geographic range of species. As part of the Missouri Resource Assessment and Monitoring Program started in 1999, the MDC has been using a stratified random sampling design to select and collect fish and physical habitat data at

100 sites across the state each year. In 2002, the Ghost shiner was collected from Elkhorn Creek in Northwest, MO and the Ozark shiner was collected at two locations within the Black River of Southeast, MO (Steve Fischer, MDC personal communication). Both species were considered locally extirpated from these watersheds because they had not been collected in these basins since the 1940's. Failure to use collections taken prior to 1970 would have excluded these watersheds from the geographic range and predictive distribution maps we produced for these two species. We can only speculate as to how many similar scenarios exist and as such it is our opinion that erring on the side of including the "historic" range of a species is preferable to excluding such areas. Essentially this is an issue of commission versus omission errors just at a larger scale. It is our contention that it is more costly, from a biodiversity conservation perspective, to exclude watersheds where a species is thought to be extirpated, but really is not (omission error) than it is to include watersheds where extirpation has actually occurred (commission error).

A second confounding factor relates to our inability to account for the effect of historic and existing human disturbances on the distribution of aquatic biota. Because satellite-derived land cover provides a depiction of the current condition of the landscape, predictions for terrestrial biota generally reflect the present-day distribution of a species, except when the species distribution is tied to unmapped or unmappable landscape features. Unfortunately, there are no satellites that provide comparable data on the present instream habitat conditions for every individual stream segment, which is the spatial grain at which our predictions are being made.

Most rivers and streams and their associated assemblages have been altered by local and watershed disturbances such as impoundments, channelization, urban and agricultural runoff, point source pollution, and the introduction of exotic species (Karr et al. 1985). Even with the significant advancements in our understanding of species-environment relations over the last 50 years, we still lack the necessary mechanistic understanding of how these and other human activities act individually or cumulatively to specifically alter instream habitat and the associated aquatic assemblages (Poff 1997). We also lack the necessary geospatial data for some of these human disturbances (e.g., channelized streams). Consequently, it is currently impossible to accurately predict the present-day distribution of the vast majority of riverine biota. The endpoint of our predictive modeling efforts for riverine ecosystems is therefore distinctly different than what can be accomplished for the terrestrial component of GAP. Due to these and other confounding factors, predictive distributions for riverine biota must reflect the biological potential (Sensu Warren 1979; Frissell et al. 1986) of a given stream segment and not the present day assemblage of species. This means that the assemblage we predict to occur in a given segment of stream will in some instances (e.g., highly disturbed streams) be quite different from the present-day assemblage. However, in relatively undisturbed locations our predictions should be relatively accurate provided our models are accurate.

4.3 General Methods

To construct our predictive distribution models we compiled nearly 7,000 collection records for fish, mussels, and crayfish and spatially linked these records to the 14-digit USGS/NRCS Hydrologic Unit coverage for Missouri and also to the stream Valley Segment coverage. Range maps were produced for each of the 315 species, sent out for professional review, and modified as needed. We then used Decision Tree Analyses to construct predictive distribution models for each species. Ultimately, a total of 571 models were developed to construct stream-reach-specific predictive distribution maps for the 315 species. The resulting maps were merged into a single hyperdistribution, which is related to a database containing information on the conservation status, ecological character, and endemism level of each species.

4.4 Detailed Methods

Selecting Species and Building a Species Occurrence Database

Through consultation with taxonomic experts we determined sufficient collection data were available to map the geographic ranges of species within four taxonomic groups: fish, mussels, crayfish and snails. We also determined that predictive distribution models would be developed for all but the snail species due to the specialized habitat requirements of this group (e.g., associations with specific water chemistries and groundwater influxes) (Wu et al. 1997). Based on a review of taxonomic references for these four groups in Missouri (Oesch 1995; Pflieger 1996; Pflieger 1997; Wu et al. 1997) a total of 366 species (32 crayfishes, 56 snails, 67 mussels; and 216 fishes) and five subspecies were selected to be included in our project. The five subspecies, all mussels, were included as distinct “data elements” at the urging of biologists both within and outside of Missouri due to concerns about the long-term persistence of these subspecies.

Data were obtained from a variety of respected data sources (Table 4.1). Obtaining collection records from the various data sources was much easier than anticipated. Most data were already in a digital format and sent to us on either floppy disks or as e-mail attachments. Collectively, we obtained over 7,000 samples with more than half being fish collections (Fish: 3,723, Mussel: 1,157, Snails: 1,086, and Crayfish: 940 samples). Collection dates range from 1900 through 1999. Except for mussels, only a few sampling biases were apparent in each database with higher concentrations of samples near locations of state and federal research or management offices and within public land holdings. The lack of sampling data for mussels in Northwest Missouri is quite pronounced, and is most likely the result of the perception that habitat degradation extirpated virtually all mussels from streams in that part of the state early in the 20th century (Oesch 1995).

Table 4.1. Data sources for collection records of each taxonomic group.

Mussels and Snails	Fish	Crayfish
• MO Department of Conservation • Dr. Ronald D. Oesch • Ohio State University • University of Missouri • University of Colorado • University of Michigan Museum of Zoology • Florida State Museum • The Field Museum of Natural History in Chicago	• MO Department of Conservation • USGS National Water Quality Assessment Program • Ozark National Scenic Riverways • Environmental Protection Agency	• MO Department of Conservation • Dr. Ronald D. Oesch

Once obtained, data were either directly transferred or manually entered into a Microsoft Access relational database developed specifically for our project. This species occurrence database consists of ten separate, but related, tables that contain three primary elements; 1) information about the collector and collection, 2) information about the location of each sample, and 3) information about the species collected. Each collection record in the database records the species present in each sample and not the actual or relative abundance of each species. The decision not to include measures of abundance was a result of many factors including; a) time and financial constraints, b) the fact that many collections did not include such information, c) the difficulty or inability to account for differing levels of effort and sampling methods among collections, and d) the inability to account for the degree of human disturbance at each sampling site at the time the collection was taken. However, to ensure a direct relationship back to the source data, we gave each collection a unique numeric identifier and inserted these unique codes into each respective source database, which allows abundance or other information not captured in our database to be retrieved by future investigators. For logistical purposes separate databases were built for each taxonomic group, but they can be easily merged into a single database in a matter of minutes.

Mapping Geographic Ranges

After completing a species occurrence database for a given taxonomic group, each collection in the database was then geographically linked, segment-by-segment (i.e., between tributary confluences), to our Valley Segment Coverage (see Section 3.7) and to the appropriate 14-digit HU within the Missouri 14-digit HU coverage using ArcView 3.2 (Figure 4.3). The nationally uniform HU system was initially developed in the mid 1970's by the USGS, Office of Water Data Coordination under the sponsorship of the Water Resources Council. This system divides the country into 21 Regions, 221 Subregions, 378 Basins, and 2,236 Subbasins (i.e., 8-digit HU's) based on surface hydrologic features (Seaber et al. 1987; FGDC 2002). In the late 1970's, the Natural

Resources Conservation Service (NRCS) initiated a national program to further subdivide 8-digit HU's into 11-digit HU's (i.e., watersheds) and then in the 1980's, several NRCS state offices, like Missouri, began subdividing the 11-digit HU's into 14-digit HU's (i.e., Subwatersheds) (FGDC 2002). *Note: The 11 and 14-digit HUs have been recently renamed and recoded as 10 and 12-digit HUs. However, during this renaming and recoding process some of the boundaries have changed. For our project we used the initial release of the 11 and 14-digit HUs, and thus retain these older naming conventions.* Each stream reach within our Valley Segment coverage and each14-digit HU has a unique numeric identifier which can be used to link tabular data, like chemical or biological data, to these spatial datasets within a GIS.

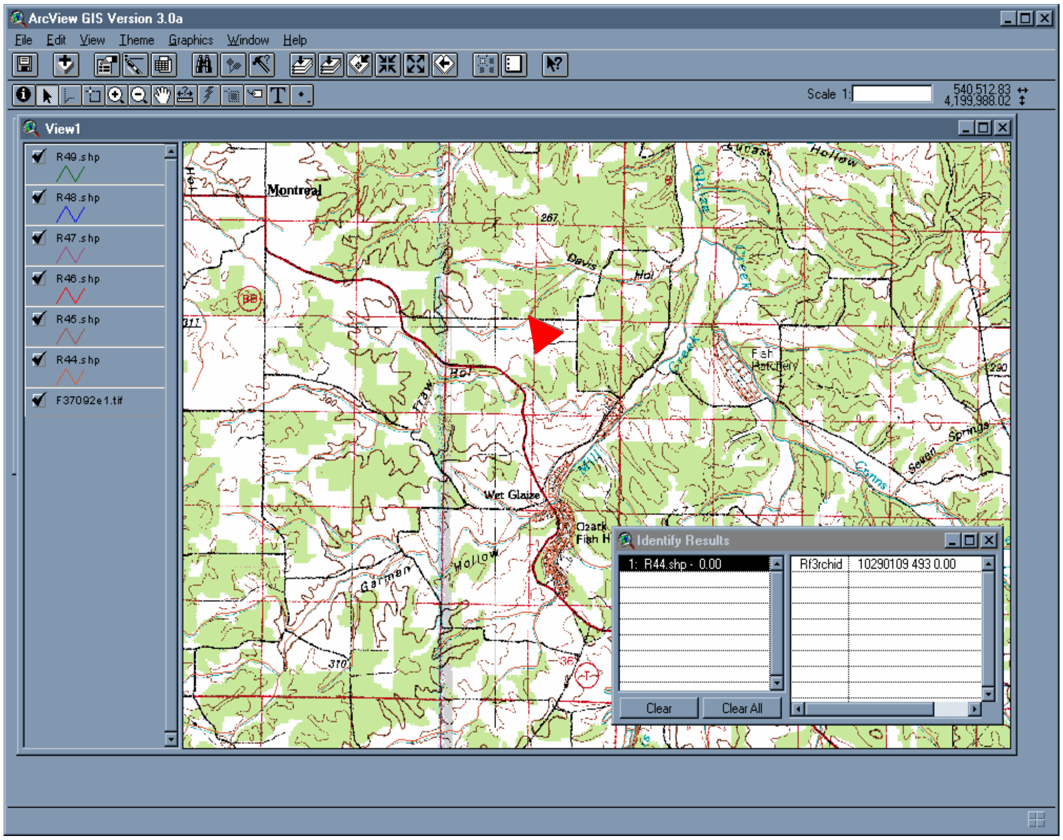


Figure 4.3. Screen capture showing an example of how unique segment identifiers were obtained for each collection record. The red arrow is pointing to a headwater stream containing a collection record and the corresponding segment id for that stream segment is displayed in the inset window. The 1:100,000 Digital Raster Graphic was used as a navigational backdrop to more efficiently identify the location of each collection.

After the segment codes and 14-digit HU codes had been obtained for every sample, we made both digital and hardcopy versions of range maps for each species for professional review (Figure 4.4). At this point we had to determine which HU coverage would be used to generate and define the geographic ranges within which our predictive models would be applied. Ideally, we would have used the 14-digit HU coverage because these represented the finest-grained spatial units available. Theoretically, as the size of the geographic unit that is used to map the range of a species decreases, the accuracy of depicting the geographic range of a species should increase resulting in lower omission and commission errors in the final predictive models (Huston 2002). This relationship only holds, however, if data are not limiting. If data are insufficient to detect all species that likely occur within all or most of the units, used to define the range of the species of interest, then the final distribution maps will be more a reflection of sampling effort than the actual distribution of species across the landscape.

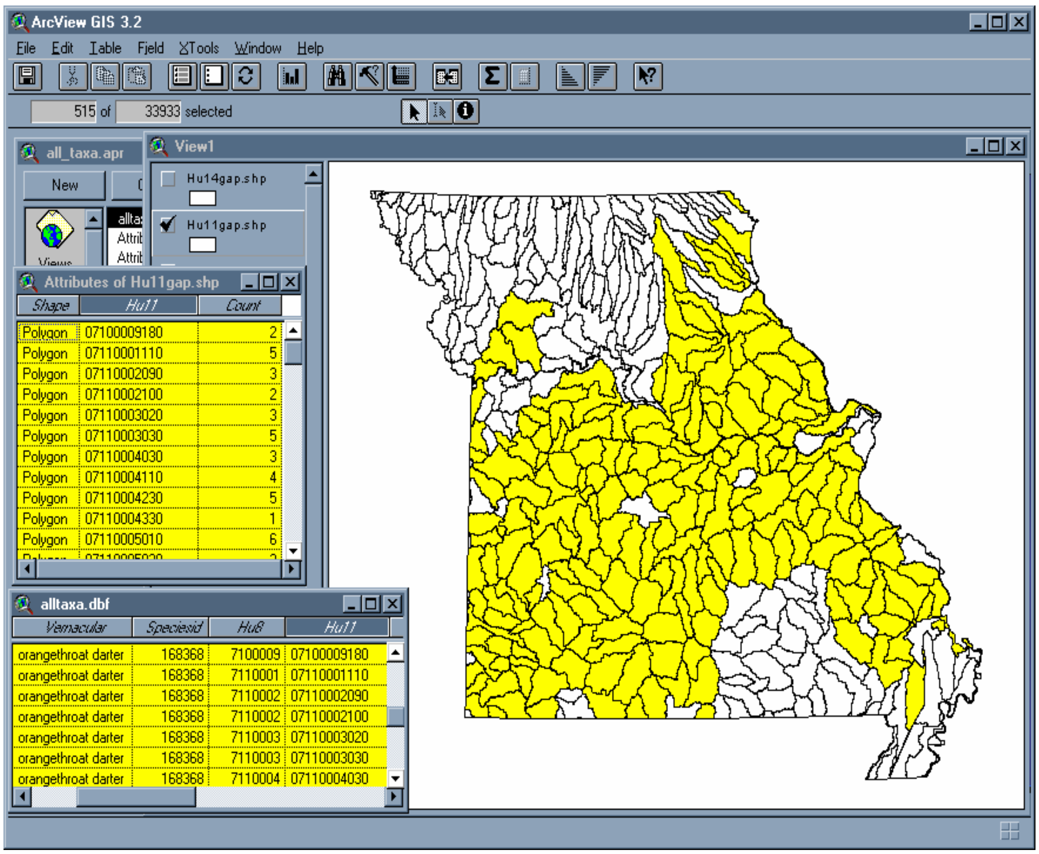


Figure 4.4. Digital version of the range map for the orangethroat darter (*Etheostoma spectabile*), by 11-digit hydrologic unit.

When using the larger 8-digit HU polygons we found that in many instances we were extending the geographic range of species outside of the true range and thus increasing errors of commission in the final predictive models (Figure 4.5). On the other hand, when using the smaller 11 and 14-digit HUs, with insufficient collection data, there was increase in omission errors in the final predictive models. Selecting the appropriate HU coverage for mapping the geographic range of riverine biota is therefore a difficult, yet very important step in the process, as it affects the overall accuracy of the final predictive models by determining the geographic extent of where your predictive models will be applied.

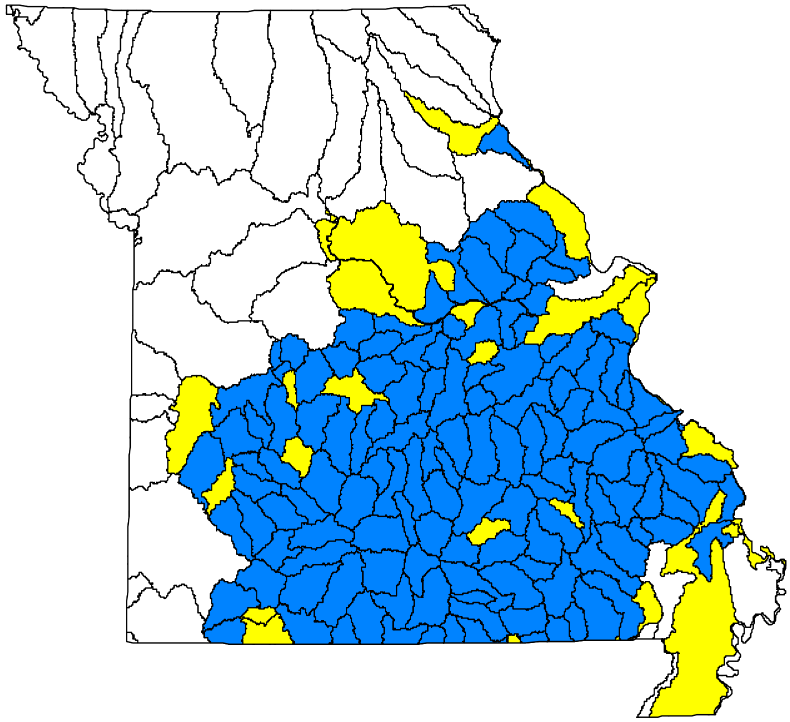


Figure 4.5. Range map of the largescale stoneroller (Campostoma) by 8 and 11-digit hydrologic unit. The range map by 8-digit HU (combined blue and yellow area) overestimates the range of this species, thus increasing errors of commission in the final distribution maps. The range map by 11-digit HU slightly underestimates the range and thus increases omission errors. For fish we decided to use the 11-digit HUs versus the larger 8 and smaller 14-digit HUs. For the other three taxa (crayfish, mussels, and snails) we elected to use 8-digit HUs due to the significantly lower number and density of samples.

There is no straightforward answer for determining the size of the spatial unit used to generate range maps for bounding where your predictive distribution models are applied across the landscape. (See Figure 4.6 to view the three hydrologic unit sizes.) It requires best professional judgment where the issues of data suitability (number and spatial coverage of collections), time, money, and the ability of experts to review maps of varying detail are all considered. When considering all of these factors we determined that for fish, geographic ranges could be generated and professionally reviewed at the 10-digit HU level. Lack of sufficient data, geographic sampling biases, and a lack of expert knowledge for many of the 11 or 14-digit HU's forced us to generate range maps for all crayfish, mussels, snails at the coarser 8-digit HU level. Although impossible to quantify, this single decision certainly increased the relative commission error rates of our predictive models for crayfish and mussels (Note: we did not develop models for snails) above those for fish, simply due to the fact that our predictive models for these taxa were more consistently applied outside of their unknown true geographic range. However, this decision also decreased our omission errors for crayfish and mussels, which as we stated above was determined to be a more desired output for the purpose of conserving biodiversity.

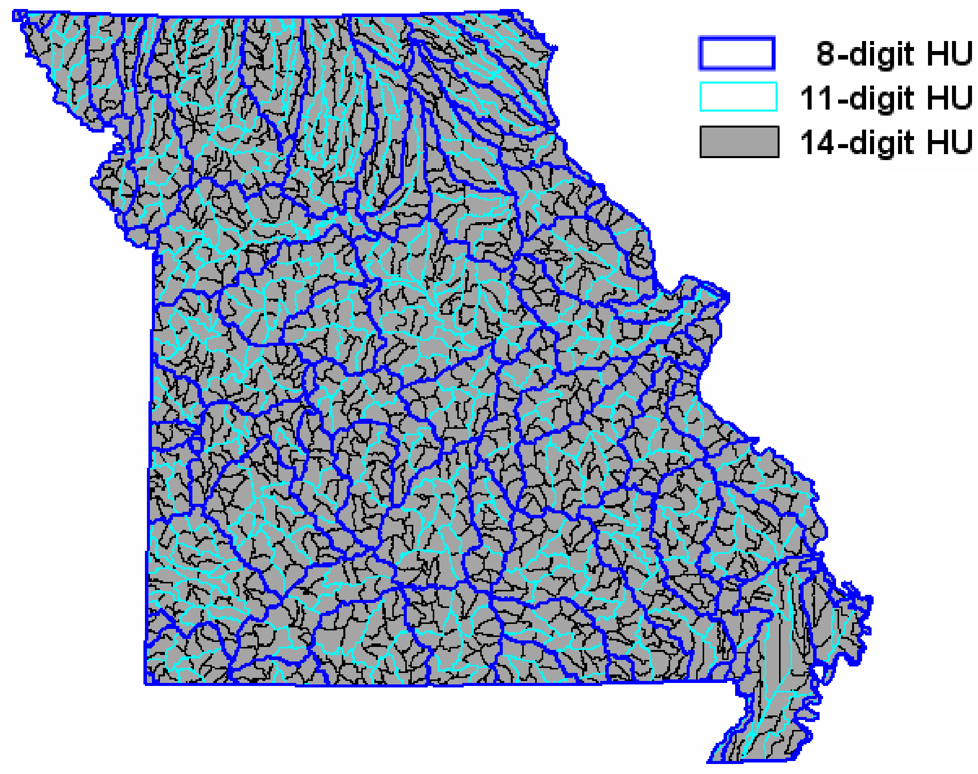


Figure 4.6. Map showing the relative size of 8, 11, and 14-digit hydrologic units.

Professional reviews of the fish and crayfish range maps were conducted on hard-copy maps, while reviews for mussels and snails were conducted on-site using digital versions of the range maps. Reviewer's edits were entered into a separate, but related, Microsoft Access database which allowed us to easily incorporate changes into the final range maps, but also keep the two information sources separate for future reviews and possible revisions. Despite our best intentions to map the geographic range of fish species as accurately as possible, it was apparent that after the professional review process was completed, the resulting database still reflected sampling biases. This was evident by the high degree of variability in species richness among adjacent 11-digit HUs.

Since we did not have the ability to conduct another professional review, yet did not want to fall back to mapping the range of fish species at the coarser 8-digit level, we devised a methodology that would still allow us to generate range maps at the more detailed 11-digit HU level. First, we constructed a series of scatter plots comparing the number of fish community samples within an 11-digit HU against the total number of species collected within each HU. To account for possible regional differences in these relations, plots were constructed separately for the OZ and CP (Figure 4.7). Both plots revealed a logarithmic relation between the number of samples and the number of species. The line of best fit or trend line also showed the typical result of such analyses where there is no discrete plateau and the relation continues toward infinity, suggesting that collections will continue to add species indefinitely. Despite this unrealistic relation both plots tend to level off at around 30 to 40 samples.

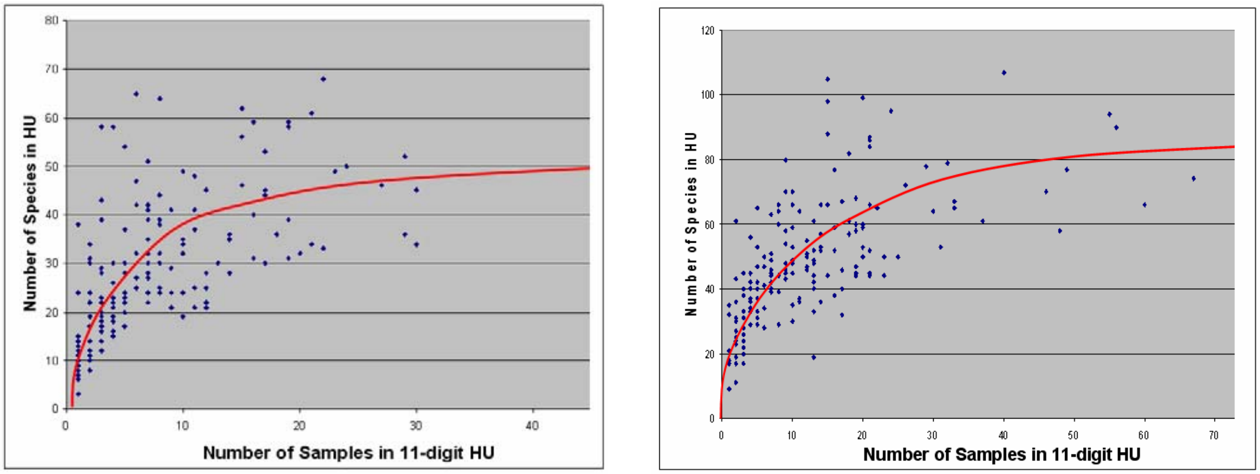


Figure 4.7. Scatter plots showing the number of samples needed to accurately census all the fish species occurring within a 11-digit HU in the Central Plains Aquatic Subregion (left plot) and the Ozarks Aquatic Subregion (right plot).

These results were quite surprising and showed that the number of collections required to accurately census all of the fish species actually occurring within a given 11-digit HU is much higher than the number of collections that presently exist for most units. However, conducting an accurate census of species within a watershed is not simply a matter of how many samples are taken. Another important factor is specifically where collections are made. Of particular interest is the size of the stream from which the collection was taken since the distribution of many riverine biota is closely associated with stream size (Huet 1959, Sheldon 1968, Strayer 1983). We therefore constructed species-by-samples plots broken out by the four stream sizes used in our valley segment classes (Headwater, Creek, Small River and Large River). These plots, which were also constructed separately for each Aquatic Subregion, reveal the same logarithmic relation between the number of samples and number of species. As expected, the trend lines leveled off at a significantly lower number of samples and suggest that approximately 10 to 15 collections are required to accurately census all of the fishes occurring within a specific stream size class of a given 11-digit HU. Yet, in every instance approximately two-thirds of the species collected with ten to fifteen samples were accounted for with just six samples. From these analyses we determined that it was reasonable to assume that most of the characteristic species for a given stream size class would be collected within an 11-digit HU when the HU contained more than five collections from that specific size class. Following this logic we developed a set of criteria for identifying “undersampled” 11-digit HU’s, which was then used to modify the geographic range of each fish species (Figure 4.8). Appendix 4.1 provides a detailed overview of this process.

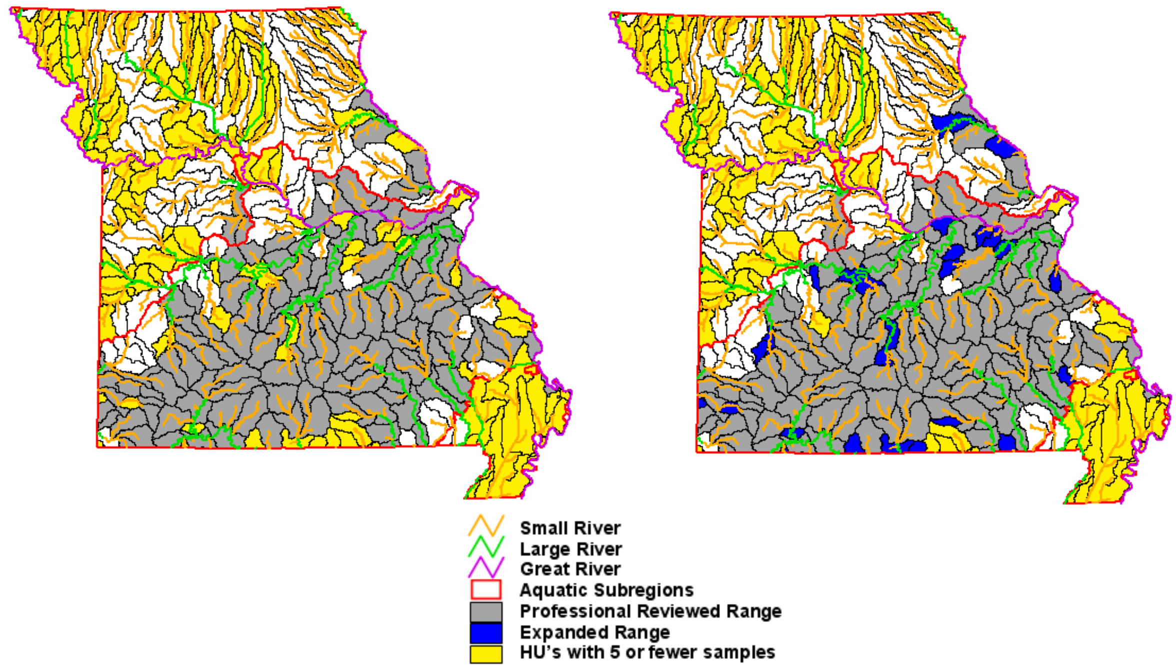


Figure 4.8. The map on the left shows the professional reviewed range of the southern redbelly dace in gray along with HUs with five or fewer samples displayed in yellow. The map on the right again shows the professional reviewed range of the southern redbelly dace (in gray) along with the HUs into which the range was expanded (in blue) because those units were determined to be undersampled (i.e., had five or fewer samples).

Constructing Decision Tree Models

<u>Statistical Methods</u>

For most species we used decision tree, also known as classification and regression tree, analyses to construct the predictive distribution models and corresponding maps. Decision tree analyses are nonlinear/nonparametric modeling techniques that typically employ a recursive-partitioning algorithm which repeatedly partitions the input data set into a nested series of mutually exclusive groups, each of which is as homogeneous as possible with respect to the response variable (Olden and Jackson 2002). The resulting tree-shape structured output represents sets of decisions or rules for the classification of a particular dataset. These rules can then be applied to a new unclassified dataset to predict which records or, in our case, location will have a given outcome.

Nonlinear models are gaining favor in wildlife-habitat relation modeling because the resulting nonparametric models define constraint envelopes of suitable habitat rather than correlations and thus more formally agree with niche theory (O'Connor 2002). That is, nonlinear models more accurately capture the normal distribution curve that species abundance will typically follow along an environmental gradient (ter Braak and Prentice 1988). Also, nonlinear models do not fall under the standard assumptions of linear, additive or multiplicative relationships, normally distributed errors, and uncorrelated independent variables, which are often unrealistic assumptions that are violated with correlative approaches (Olden and Jackson 2002; Huston 2002; O'Connor 2002). Decision trees, in particular, have become a popular modeling technique because they construct models with accuracy comparable to the more "sophisticated" nonlinear methods (e.g., Neural Networks; Olden and Jackson 2002), and yet are much easier to construct and interpret (Breiman et al. 1984; De'ath and Fabricus 2000).

We used AnswerTree® 3.0 to construct all of our decision tree models. AnswerTree® is an extension of the SPSS® statistical software package, which brings together four of the most current and widely used analytic methods or algorithms for performing decision tree analyses; a) Chi-squared Automatic Interaction Detector (CHAID), a method that uses chi-squared statistics to identify optimal splits (Kass 1980), b) Exhaustive CHAID which is a modification of CHAID that does a more thorough job of examining all possible splits for each predictor but takes longer to compute (Biggs et al. 1991), c) Classification and Regression Trees (C&RT or CART), methods that are based on minimization of impurity measures (Breiman et al. 1984), and d) Quick, Unbiased, Efficient Statistical Tree (QUEST), a method that is quick to compute and avoids other methods' biases in favor of predictors with many categories (Loh and Shih, 1997). The four algorithms all perform basically the same thing; examine all of the predictor variables in your database to find the one that initially gives the best classification or prediction of the target variable by splitting the data into subgroups (AnswerTree® 3.0 User's Guide 2001). The process is then applied recursively to subgroups to define sub-subgroups, and so on, until the decision tree is completed, as determined by user-defined stopping criteria.

To help decide which of the four algorithms to use we used all four methods to construct models for a handful of species. We then compared the efficiency, consistency, and reliability of the four algorithms and also how easily each of the outputs could be interpreted. These comparisons showed that all four methods were very efficient, taking only seconds to analyze even the largest input dataset (approx 2300 collection records, with 7 potential predictor variables). The different methods also consistently constructed the same models and all appeared to be reliably synthesizing and segmenting the data into meaningful subgroups. This determination was made by examining the general correspondence of the resulting models with the habitat-affinity information extracted from the literature and the contingency tables produced independently for each predictor variable. The only major difference between the methods pertained to interpretation of the output. Both C&RT and QUEST are binary-tree growing algorithms, meaning that at each split only two subgroups or nodes are generated. Consequently, these methods tend to grow trees with many levels (long trees) where the same predictor variable is often used over and over to split the tree into a number of different successive levels (AnswerTree® User's Guide 2001). CHAID and exhaustive CHAID on the other hand are not binary and can produce more than two categories at any particular level in the tree (Kass 1980; Biggs et al. 1991). These methods tend to create wider trees than the binary growing methods. We found the wider trees of CHAID and Exhaustive CHAID much easier to interpret and since we found no other differences in performance among the four algorithms we decided to use Exhaustive CHAID to construct our models.

Exhaustive CHAID is a modification of CHAID developed by Biggs et al. (1991). It was developed to address some weaknesses of the CHAID method. In some instances CHAID may not find the optimal split for a variable since it stops merging categories as soon as it finds that all remaining categories are statistically different (AnswerTree® User's Guide 2001). Exhaustive CHAID remedies this problem by continuing to merge categories of the predictor variable until only two "supercategories" are left and then examines the series of merges for the predictor and finds the set of categories that gives the strongest association with the target variable and computes an adjusted-*p* value for that association. Consequently, exhaustive CHAID can find the best split for each individual predictor and then choose which of these predictors to split on at each level in the tree by comparing the adjusted-*p* values.

GIS Base Layer for Predictive Modeling

The base layer for our predictive distributional modeling efforts was our Valley Segment Type (VST) coverage, which represents a substantially edited and enhanced version of the 1:100,000 NHD. The finest resolution ("linear spatial grain") of our predictions was the stream segment, which in most instances is represented by a section of stream between tributary confluences (i.e., analogous to a city block). Deviations in this spatial definition arise when changes in the temperature, flow, or feature type (i.e., waterbody vs. stream) occur somewhere between confluences. Within the state boundary of Missouri there are approximately 106,000 individual stream segments within our 1:100,000 VST coverage. These segments have an average length of 1.7 Km.

Classification/Predictor Variables

Animals live where there is sufficient local availability and proper spatial arrangement and connectivity of critical resources that correspond to morphologically, physiologically, and behaviorally-mediated ecological requirements that determine survival and reproductive success (Wiens 1989; Schlosser 1995; Matthews 1998). Of particular importance is that species use resources at multiple spatiotemporal scales and that resource availability is determined by processes operating at multiple spatiotemporal scales (Smith and Powell 1971; Frissell et al. 1986; Jackson and Harvey 1989; Tonn 1990; Rabeni and Sowa 1996; Poff 1997; Jackson et al. 2001). Potential distributions are often constrained by isolation mechanisms and by biotic interactions, particularly predation (Jackson et al. 2001) and competition (Winston 1995) or also in the case of most freshwater mussel species, host-parasite relationships (Parmalee and Bogan 1998). As a consequence of these constraints species rarely, if ever, occupy all suitable locations (Hutchinson 1957).

Under ideal circumstances, predictive models should be based on mechanistic understandings of functional relationships between life history requirements and abiotic and biotic selective forces operating at multiple spatial and temporal scales (Poff 1997; Jackson et al. 2001; Maurer 2002). Poff (1997) provides a valuable heuristic framework for developing hierarchically-nested mechanistic models to predict the distribution and abundance of riverine biota and we certainly agree that we must strive to achieve such predictive capabilities. However, upon examining even the most basic information required for practical application of mechanistic models within a GIS, it becomes readily apparent that much of the necessary information is currently lacking (Maurer 2002). First, we need to know which life-history traits (e.g., foraging or reproductive strategy, thermal tolerance, etc...) are most subject to natural selection (Orians 1980; Poff 1997) and how to best categorize these traits into ecologically meaningful response groups (e.g., reproductive guilds; Balon 1975). Upon addressing these issues we must then have sufficient life history data to allow <u>all</u> species of interest to be accurately placed into the appropriate response groups. We must also identify which abiotic and biotic factors serve as critical selective forces at multiple spatial and temporal scales (e.g., climate, geology, temperature, substrate, predation, competition), the interactions among these factors over space and time, and then establish functional relations between response groups and selective forces and their interactions (Wiens 1989; Poff 1997; Cushman and McGarigal 2002).

Poff (1997) convincingly shows that we have made significant advancements in identifying and categorizing important species traits and selective forces and even establishing functional relations between these factors. What is lacking is sufficient life history information for many species that would allow accurate placement of species into functional response groups, especially when you consider ontogenetic shifts in ecological requirements (Matthews 1998; Maurer 2002). An even greater hindrance to mechanistic approaches to modeling within a GIS pertains to the lack of detailed geospatial data on critical selective forces. Many of the variables that potentially act to directly or indirectly constrain the distribution or determine the abundance of riverine biota (e.g., substrate composition, thermal regime, flow regime) either have not been or cannot be mapped across broad areas within a GIS at "reasonable" spatial scales (e.g., 1:100,000 or 1:24,000). Present efforts to map the predictive distribution of riverine biota across the landscape must therefore rely on mappable proximate variables, which are generally associated with a variety of specific selective forces.

In the study of stream fishes, four factors have proven most effective for predicting patterns of distribution and abundance: measures of stream size, gradient, temperature regime, and flow regime (Moyle and Cech 1988). We selected seven variables as potential predictors, all of which specifically or generally pertain to these four factors (Table 4.2). In addition to their "reputation" as useful predictors, these variables were selected because they either already had been or could be mapped within a GIS at a scale of 1:100,000. The specific details of how these variables were mapped and attributed to each individual stream segment within our VST coverage are described in Section 3.7.

Table 4.2. Predictor variables used to model fish, mussels, and crayfish. Five variables were chosen for each model, one from each variable type.

Variable Name	Full Name	Predictor Variable Type
Size	Stream Size	Stream Size Measure
LinkR	Shreve Link Range	Stream Size Measure
GradSegR	Range of Stream Segment Gradients	Gradient Measure
RGradSub	Relative Gradient by Subregion	Gradient Measure
Temp	Stream Temperature	General StreamTemperature
Flow	Stream Flow	Constancy or Permanence of Flow
Sdisc_2C	Size Discrepancy Two Class (yes or no)	Size Discrepancy

Stream Size Measures

It has long been recognized that a wide array of structural features and functional processes occurring within and along stream ecosystems tend to change in a longitudinal continuum from the smallest headwaters to the largest rivers (Vannote et al. 1980). This continuum of change is certainly not a universal truth as there are minor exceptions resulting from local factors (e.g., lakes or springs), which create discontinuity in the continuum. It is also important to note that longitudinal changes in hydrogeomorphic and physicochemical character tend to be regionally-specific (Minshall 1978; Naiman et al. 1987). However, given the general consistency of longitudinal change in environmental conditions within a given physiographic region (See Minshall et al. 1983) it is not surprising that numerous investigators have identified an associated continuum of change in biological assemblages (See Matthews 1986 for a review).

Instead of using the more precise measures of drainage area or discharge most investigators have utilized discrete stream size classes (Sensu Horton 1945 and Strahler 1957) in order to more tractably investigate and communicate longitudinal changes in the abiotic and biotic character of streams. The Strahler ordering system is certainly the most widely recognized and the one most often used by stream ecologists for research and management (Hansen 2001). References to *Strahler order* tend to dominate habitat-affinity descriptions of riverine biota in the general literature and especially in taxonomic texts. However, *Strahler order* often underestimates stream size due to vagaries in drainage network structure (Hynes 1970). With the Strahler ordering system it is common to have lower order streams with substantially larger drainage areas than higher order streams. Recognizing this problem Shreve (1966) devised another measure of stream size, termed *link magnitude*, which largely overcomes this problem since it is much more precisely related to drainage area (Hansen 2001).

We used two different measures of stream size as potential predictor variables. Both are based on Shreve *link magnitude* (Shreve 1966). The first measure of stream size directly corresponds to the five size classes that were used to classify valley segment types; headwater, creek, small river, large river, and great river (see Section 3.7). The second measure, which for internal data management purposes we simply labeled *LinkR*, breaks each of these five generalized categories into two subcategories. This *LinkR* measure of stream size is therefore twice as precise (10 vs. 5 categories) at categorizing stream size. The reason we used two different measures of stream size related to the fact that we wanted to maximize the accuracy and precision of our predicted distribution maps. Ideally, we would have preferred using the more precise *LinkR* measure as the potential predictor for all species. However, this variable was often not suited for species with a limited number of occurrence records. In these situations, occurrence percentages tended to dramatically fluctuate among the more precise *LinkR* categories simply because a small change in the number of occurrences in any given category resulted in a relatively large change in the occurrence percentage. In many instances these fluctuations would obscure an obvious distributional association with stream size and the variable would not be retained as a predictor in our decision tree model. Replacing the *LinkR* measure with the more general *Stream Size* classes usually rectified this problem because more collections fell within each category and the relative influence of a single occurrence record on the occurrence percentages was substantially reduced.

<u>Size Discrepancy</u>
Size discrepancy, another predictor variable included in our decision tree analyses, provides a general measure of the position of each stream reach within the larger drainage network. In our coding scheme a 0 (zero) indicated no stream size discrepancy while a code of 1 indicated that there was a size discrepancy. This binary variable is based on our five stream size classes of headwater, creek, small river, large river and great river. A code of 0 (zero) designates reaches that flow into a reach of the same size category (e.g., headwater flowing into another headwater) while a code of 1 designates reaches connecting to a reach falling into a larger size category (e.g., headwater flowing into a small river, large river, etc...). Several investigators have found that fish assemblages in the lower sections of streams are often influenced by the size of the confluent stream (Fowler and Harp 1974; Gorman 1986; Osborne and Wiley 1992). We also noticed similar patterns in our dataset where “small or large-river species” were often collected in reaches of headwaters and creeks that directly connected to these larger streams. There is no way of knowing whether or not these larger river species are using the lower ends of smaller tributaries to fulfill any critical life-history function, however, the consistency of the pattern suggested that it would be better to include rather than exclude this variable from our modeling efforts. When the size discrepancy variable is included in one of our predictive models it simply reveals that that particular species was consistently found in the lower reaches of smaller tributaries that directly connect to the larger streams where that species more typically occurs.

<u>Gradient Measures</u>
Gradient or channel slope has long been recognized as a principle adjustable property of rivers that is often found to be associated with the distribution of riverine biota (Huet 1959). Every river strives to establish a quasiequilibrium state in which several

interdependent variables (width, depth, velocity, gradient, and hydraulic roughness) are adjusted to accommodate the dominant discharge (typically bankfull) and sediment load (Ritter et al. 1995; Jacobson et al. 2001). Gradient is therefore largely determined by discharge and sediment load and not surprisingly has been shown to be associated with the related variables of mean annual discharge, drainage area, and median size of bed material (Hack 1957; Nino 2002). Because sediment loads vary geographically according to geologic, soil, and landform controls these relations are typically only revealed when examined within a given geologic or physiographic setting (Hack 1957). Within such a context gradient is positively correlated with median substrate particle size and is negatively correlated with drainage area or stream size (Hack 1957; Nino 2002). At segment, reach, and local scales, gradient influences water velocities and shear stresses and can also influence mean values and spatial patterns of stream temperature and dissolved oxygen concentrations via its influence on mixing within the water column and enhancing gas exchange between the water surface and the overlying air column (Moyle and Cech 1988; Knighton 1998). Gradient can also be associated with the presence, character, and diversity of habitat types (e.g., pools, riffles and runs) (Moyle and Cech 1988). This relation between gradient and so many key environmental variables is what makes this factor such a potentially useful predictor of species distributions within riverine ecosystems.

We used two very different gradient variables as potential predictors. The first variable, labeled GradSegR, represents actual stream segment gradients broken into ten equal interval categories for modeling purposes. As discussed above, within each Aquatic Subregion, this variable is negatively correlated with drainage area or stream size. However, since sediment load and median bedload particle sizes vary among our Subregions, this association with drainage area does not hold for a statewide perspective and Ozark streams, with their relatively coarse substrates, tend to have significantly higher gradients than the other two Subregions for any given stream size (Pflieger 1971).

This GradSegR variable often proved very informative and served as an accurate predictor for some of the species that occur in both the CP and the OZ. Typically, it was necessary to develop regionally-specific models for species that occur in these two Subregions. However, for species that mainly occur in one of these two regions and only peripherally in the other, a single and more accurate multi-region model could be developed using GradSegR. These trans-regional relationships with GradSegR possibly result from a species’ association with a particular range of substrate conditions. This conclusion is certainly based on the assumption that the relation between gradient and substrate composition within a Subregion also remains intact along the periphery of that Subregion. This assumption is generally supported by comparative descriptions of stream habitat conditions within the transitional Ozark Border region that separates the Ozark and Central Plains Aquatic Subregions (Pflieger 1997).

The other gradient measure, RGradSub, is a relativized measure of stream gradient that categorizes each stream reach as being either; low, intermediate, or high gradient. These categories are relative to each stream size class and Aquatic Subregion.

However, this variable does not apply to streams in the Mississippi Alluvial Plain Subregion since stream gradients in this Subregion are too low (typically less than 1 foot per mile) to allow accurate characterization and categorization using a 30-meter DEM.
The RGradSub categories account for and largely remove the significant association between gradient and drainage area in order to better identify local scale variations in substrate conditions, water velocities, and the availability of habitat types. With this variable we should expect that, within a given size class and Subregion, as you move from the low to the high-gradient category you will find relatively coarser substrates, higher velocities and higher diversity of habitat types. Another way of looking at this variable is that the lower gradient classes of a given stream size category will likely have instream habitat conditions similar to those found in the next larger size class. In fact, we often found species that primarily occur in larger size classes also had high occurrence percentages for the low gradient categories of the next smaller size class. For example, the decision tree model for the shortnose gar within the Ozark Aquatic Subregion selected all LinkR categories greater than or equal to 7 (i.e., essentially large rivers) and also LinkR categories of 5 and 6 (i.e., larger small rivers) that were further classified as having relatively low gradients.

Temperature

Stream ecologists have long recognized the important role that temperature plays in determining the distribution of riverine biota (Huet 1959). The prominent influence of temperature on the distribution and abundance of aquatic organisms is not surprising considering the diverse and pronounced effect temperature has on the reproduction, growth, behavior, physiology, condition, and survival of ectothermic animals (Ferguson 1958; Magnuson et al. 1979; Reynolds and Caterlin 1979).

Even relatively small temperature differences can result in dramatic differences in aquatic community composition (Karr and Schlosser 1978; Matthews 1987; Binkley and Brown 1993; Jacobsen et al. 1997; Rabeni et al. 1997b). Rabeni et al. (1997b) found maximum summer stream temperatures to be highly correlated with composition and relative abundance of fish species inhabiting headwater streams of an Ozark watershed. As maximum summer temperature increased species with higher, laboratory-determined, hyperthermia tolerance values were either added to the community or increased in relative abundance. In essence, this study showed that with each 1 °C change in maximum summer temperature there was a corresponding change in fish community composition. Temperature minima and thermal variability can also influence the distribution and abundance of riverine biota since sustained low temperatures cause metabolic stress on ectothermic animals (Cunjak 1988) and severe fluctuations in temperature can lead to direct thermal shock to eggs and fry or cause changes in reproductive behavior that lower reproductive output (Shuter et al. 1980).

Like many states across the US, Missouri’s streams exhibit a wide range of thermal conditions even within a given stream size class (Smale and Rabeni 1995b; Sowa and Rabeni 1995). Maximum stream temperatures can range anywhere from 15 to 30+ ° C throughout Missouri (Scott Sowa, personal observation). This spatial variation in stream temperatures is especially prevalent in karst regions like the Ozark Aquatic Subregion where largely unpredictable groundwater inputs, mainly in the form of diffuse and conduit springs, result in a heterogeneous matrix of thermal conditions that defy efforts to model stream temperatures across the landscape (Pflieger 1971; Vineyard and Feder 1979).

Despite the ecological importance of temperature and the wide range of thermal conditions across the state, we currently lack any sort of detailed spatially continuous map of stream temperatures for Missouri. The only geospatial data available for discriminating among stream temperatures was a "coldwater streams" datalayer produced by the Missouri Department of Conservation. This coldwater streams datalayer is certainly not comprehensive and largely corresponds to those stream segments supporting naturalized trout populations or put and take trout fisheries. We used this datalayer to classify the appropriate stream segments in our 1:100,000 baselayer as "cold" and all other stream reaches were simply classified as "warm". Despite the spatially restrictive or incomplete nature of this coverage and the extremely generalized temperature categories, this binary temperature variable did prove to be a surprisingly important predictor of species distributions.

Constancy or Permanence of Flow

Constancy of flow (i.e., perennial vs. intermittent) during normal annual low-flow conditions can have a dramatic influence on the composition and abundance of riverine assemblages. Intermittent streams are characterized by harsh environmental conditions, such as low dissolved oxygen and high temperatures, and often intense competitive and predator-prey interactions (Matthews 1987; Matthews 1998). These harsh conditions and intense biotic interactions result in a fairly restricted fish and mussel assemblage comprised by species that have evolved behavioral strategies or physiological tolerances to deal with such conditions (Neel 1951; Larimore et al. 1959; Moyle and Cech 1988; Matthews 1987; Pflieger 1997). It is suspected that adaptations (e.g., burrowing) by crayfish allow them to overcome these harsh conditions and that the lowered predation risk from large predatory fish generally found within intermittent streams, enable crayfish populations to reach significantly higher densities than in perennial streams (Flinders and Magoulick 2003).

Intermittent flow has been variously defined, but generally refers to stream channels that have no surface flow for part of the year (Pflieger 1989; Hansen 2001). In Missouri, intermittent streams are often characterized by a series of isolated pools separated by riffles either completely or nearly lacking surface flow during summer base-flow conditions. For much of the remainder of the year these channels will contain continuous surface flow.

Intermittent stream reaches are given a distinct code within the NHD and most other reaches are simply considered to be perennial. These existing attributes were used to designate the two constancy of flow categories (perennial vs. intermittent) used in our predictive models. The majority of intermittent reaches in the NHD and thus our VST coverage are headwater streams, which is reasonable considering that constancy of flow is related to drainage area in a regionally-specific context (Matthews 1987; Hansen 2001). However, as a result of the karst landscape of the Ozarks a number of larger streams or rivers in this region lose a substantial portion, or sometimes all, of their flow to the underlying groundwater system (Vineyard and Feder 1979). These so called "losing streams" may be either completely dry or contain a series of isolated pools under normal baseflow conditions.

Distinctions between perennial and intermittent channels within the NHD are certainly not without error. In fact, these errors can be quite large as Hansen (2001) found in a comparative analysis conducted in the Chattooga River watershed within the southeastern US. These errors can arise from various sources. For instance, the original intermittent flow designations were made via interpretation of aerial photos, which is certainly subject to observer error. Also, for streams covered by a dense riparian canopy only speculation can be used to determine flow conditions. Errors also result from the fact that the aerial surveys were conducted during various times of the year and not specifically annual low-flow conditions. Unfortunately, the prevalence of these errors is unknown and can only be quantified through intense ground verification (Hansen 2001).

Generating Input Datasets for Each Species

Generating the input datasets (i.e., suite of collection records) to be used for constructing predictive models is an important step in the overall modeling process. First, you have to ensure that a species actually has the ability populate all collection locations included in the input dataset. Failure to do so could alter model parameters and accuracy or even eliminate the possibility of constructing a statistically significant model. As evidenced by the successful establishment of numerous species outside of their native geographic range, it is quite obvious that species rarely occupy all suitable habitats across the landscape (Krebs 2001). This is especially true for obligate aquatic biota where certain distributional constraints, like dispersal barriers or time since last disturbance (e.g., glaciation) play such a prominent role in determining the geographic range of a species (Hocutt and Wiley 1986; Mandrak 1995). Because these distributional constraints will act to decouple species-environment relations and thus hinder model development (Wiens 1989) we only used collection records within the 8-digit HUs from which a given species had actually been collected to define the input dataset for each species.

There is another issue to consider when defining which collection records will be included in the model-building effort, especially when you are using proximate predictor variables. This second issue pertains to the fact that species often exhibit regional variation in their relation with predictor variables. For instance, Pflieger (1997) notes that the shorthead redhorse (*Moxostoma macrolepidotum)* is the most abundant redhorse in downstream sections of the largest Ozark rivers, however, in the Central Plains it frequents much smaller streams. Such regional variations could result from; 1) regionally specific biotic interactions causing niche displacement, 2) genotypic variation across the species complex resulting in geographic variation in the fundamental niche, or 3) suitable environmental conditions occurring at different locations (e.g., stream size class) within the overall stream network. This last cause could be ignored if we were able to use the full suite of the actual potential limiting factors as our predictor variables. However, when using proximate predictor variables, like stream size and gradient, we have to take this variation into consideration no matter what the cause of the variation.

Accounting for regional variation can be accomplished in two ways, either include "region" as a potential predictor or construct regionally-specific input datasets. We used the second option since there were often gross imbalances in the number of occurrence records among our Aquatic Subregions, which often led to the significant relations found in the Subregion with the most occurrence records overriding subtle regional variations in the associations with an important predictor like stream size. So, for those species that occurred in both the Plains and the Ozark Aquatic Subregions we developed multiple input datasets. Specifically, were generated one combined dataset, which included all of the collection records within the 8-digit HUs from which the species was collected in both Subregions, and then separate datasets for each Subregion. We constructed models for all three datasets and then selected either the combined model or the regionally-specific models to generate our predictive distribution maps based on which one(s) yielded the highest predictive performance.

It is important to note that regionally-specific datasets were only constructed and used for possible model development when there were at least 30 occurrence records for a given species within that Subregion. This cutoff criterion was based on our personal experience with initial modeling trials that showed that ~30 occurrence records were needed for generating any sort of reliable model. It is also important to recall we were unable to construct decision tree models for the MAB Subregion because we did not have the same suite of predictor variables that we had for the other two Subregions. For this reason, we never constructed statewide input datasets and had to simply rely on general associations with stream size categories to construct predictive distribution maps within the MS Alluvial Plains Subregion.

Model Selection Criteria

<u>Stopping Criteria</u>

AnswerTree® 3.0 allows the user to specify apriori stopping criteria related to the size of the tree (i.e., number of levels) and the minimum number of collection records that can occur in any given child node. These stopping criteria help reduce the probability of gross overfitting of the model which can be a problem with extremely large datasets containing a large number of predictor and/or response variables (Answer Tree® User's Guide 2000). Because we had a limited number of predictor variables and also developed efficient pruning criteria, overfitting of the models was not a major concern. We set the maximum number of levels allowable in the final tree equal to 10, which was higher than the number of levels ever achieved. We set the minimum number of collections allowable in a child node equal to 1. We did not specify a higher number of cases because we found that the habitat-affinity information compiled from the literature could be used to assess the general validity of child nodes with a low number of collections.

Growing Criteria

We set the alpha level for splitting and merging equal to 0.1 and used the Bonferoni alpha adjustment to account for the increased likelihood of a Type One error associated with the multiple comparisons. This more liberal alpha level was used because initial modeling trials revealed that the more conservative alpha of 0.05 sometimes failed to generate a model altogether, but more often it would generate a highly restrictive model. We then increased the alpha level to 0.2 if no model resulted at an alpha of 0.1, but we never increased the alpha level above 0.2. If no model was generated at an alpha of 0.2 we used habitat-affinity information obtained from the literature in conjunction with contingency tables of the predictor variables and histograms showing a species frequency of occurrence by VST to generate a more subjective predictive model. If no model could be generated using any of these methods, the final distribution map for that particular species simply represents the stream reaches where the species has actually been collected.

Pruning Criteria

The recursive partitioning algorithms used in decision tree analyses tend to overfit the data and produce trees with too many levels and terminal nodes (Breiman et al. 1984). When ovefitting occurs the user should “prune” the decision tree by evaluating model performance under different tree structures and then remove nodes that actually increase misclassification rates or otherwise negatively alter other measures of model performance (e.g., sensitivity or specificity; see Olden and Jackson 2002). Although simple in theory, such evaluations are very time consuming and inefficient. With 571 models to evaluate we had to develop efficient pruning criteria in order to select which of the nodes from the overfit models would be used in the final model(s) for each species. We decided to use a “relative 50%-approach” to select which nodes to include in each final model. For each model we first identified the node with the highest occurrence percentage that also contained at least 5% of all the collection records from the overall input dataset. For example, if the input dataset contained 1000 total collection records, we would identify the highest occurrence percentage for those nodes having 40 or more total collection records. This was done to account for the fact that terminal nodes with only a handful of samples generally provided grossly inflated or deflated occurrence percentages. We then divided the highest occurrence percentage by 2 and selected all nodes having occurrence percentages greater than or equal to this percentage. For example, if the highest occurrence percentage was 80% we would select all nodes with occurrence percentages greater than or equal to 40% and if the highest occurrence percentage is 50% we would select all nodes with occurrence percentages greater than or equal to 25%. Figure 4.9 shows an example of this process for the wedgespot shiner. This proved to be an efficient and standardized approach that accounted for differences in species prevalence or commonness, which were not accounted for by the other model evaluation tools included in Answer Tree 3.0.

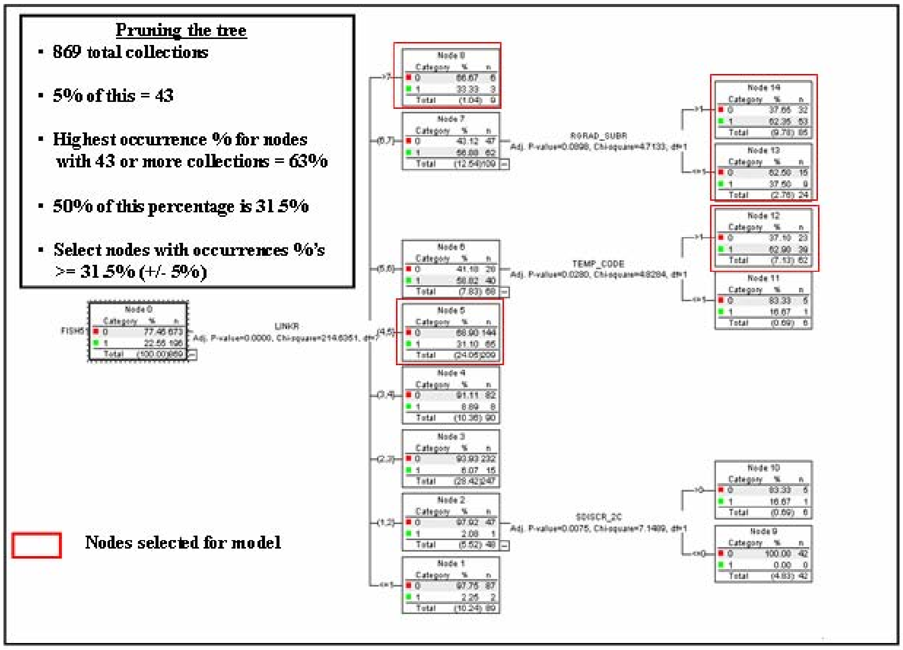


Figure 4.9. Example of a decision tree for the wedgespot shiner and the pruning criteria used to generate the final predictive model.

These pruning criteria led to the pattern of commission and omission errors that is often found in wildlife habitat modeling (Karl et al. 2002). Lowest commission and highest omission errors consistently occurred for species that could be generally categorized as having broad geographic ranges, being common throughout the range, typically abundant wherever suitable habitat exists, and having relatively good dispersal capabilities (e.g., many Centrarchid species). Essentially these are common species with high detectability and that are likely to stray into, and thus be collected in, marginal or unsuitable habitats. Highest commission and lowest omission errors, on the other hand, consistently occurred for species categorized as having either narrow or broad geographic ranges, being patchily distributed throughout the range, rarely or never abundant even within areas of suitable habitat, and relatively poor dispersal capabilities (e.g., many darter and minnow species). These are rare or patchily-distributed species with low detectability that are seldom collected outside of areas of suitable habitat.

4.5 Results

Statewide distribution models and maps were developed for 216 fish, 67 mussel, and 32 crayfish species. The number of models generated for any given species ranged from one to four, with most species requiring two regionally-specific models to account for regional variations in habitat associations. In all, we developed a total of 571 models in order to generate the predicted distribution maps for all 315 species. The resulting predictive distribution models, maps, and the habitat affinity information compiled from the literature for each of these species are provided in Appendices 4.2 - 4.4.

Species Richness

Species richness is just one of many measures of biodiversity, and while our interest is ultimately on assessing the representation of species and all distinct riverine ecosystems within the existing matrix of public lands, patterns of species richness are an interesting element of biodiversity. Considering all 315 species included in our project, our models collectively predicted the richest stream segments to contain 146 species, 46% of the total (Figure 4.10). The highest richness values for stream segments among taxonomic groups contained 47% of 216 fish species, 61% of 67 mussel species, and just 22% of 32 crayfish species (Figures 4.11-4.13). The relatively low percentage for crayfish is a result of the fact that many of the crayfish species in Missouri have relatively limited geographic ranges, often found in just a single or a handful of drainages (see Figure 4.13).

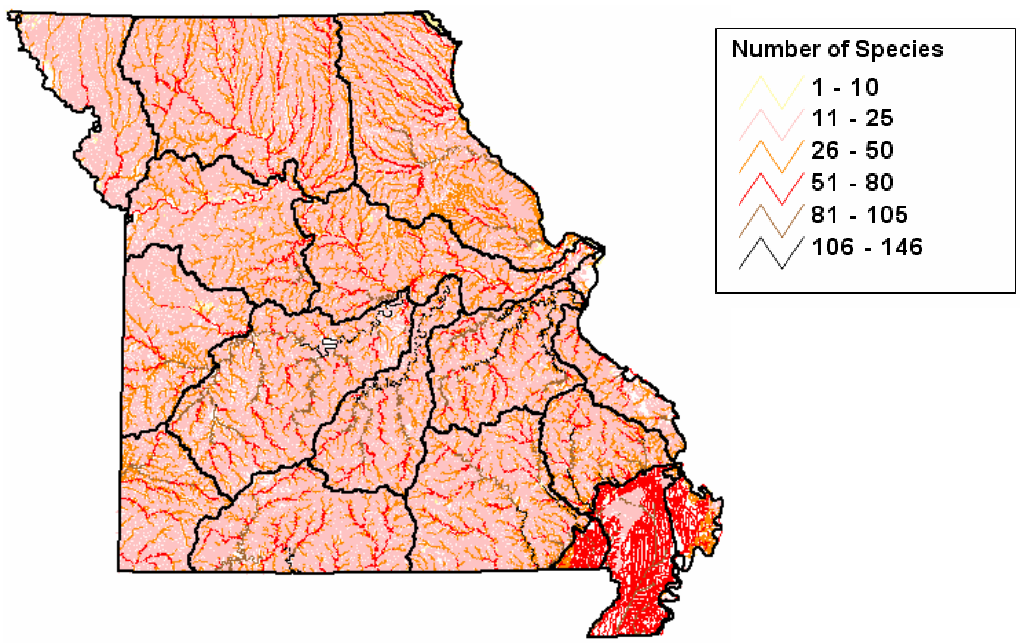


Figure 4.10. Map showing overall species richness (fish, mussel, and crayfish) by stream segment.

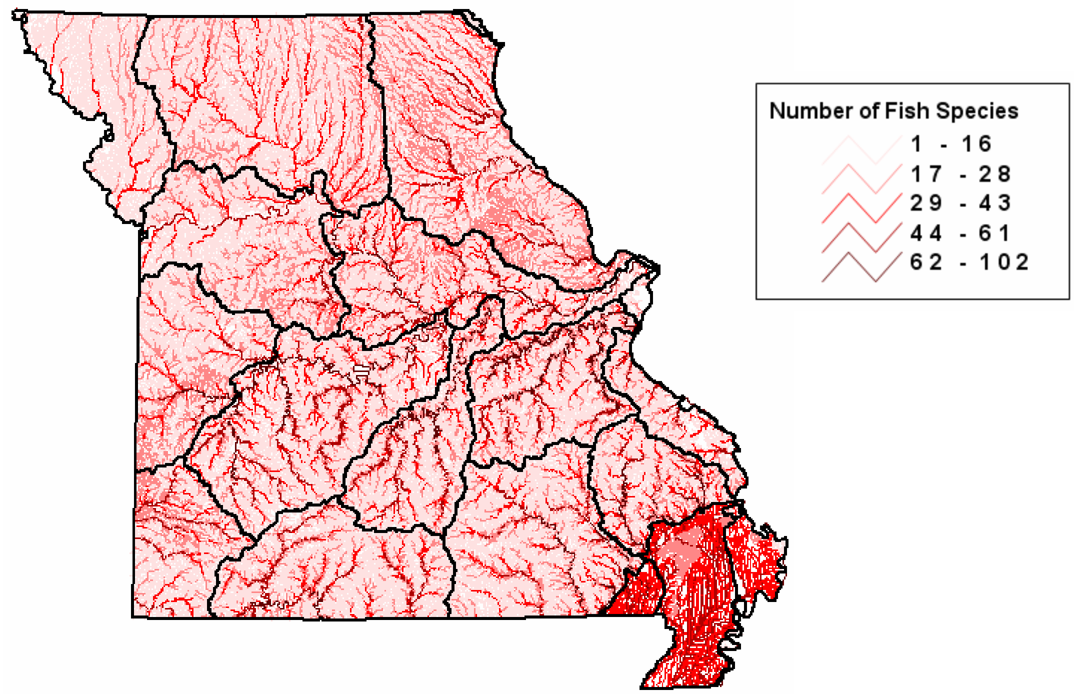


Figure 4.11. Map showing predicted fish species richness by stream segment.

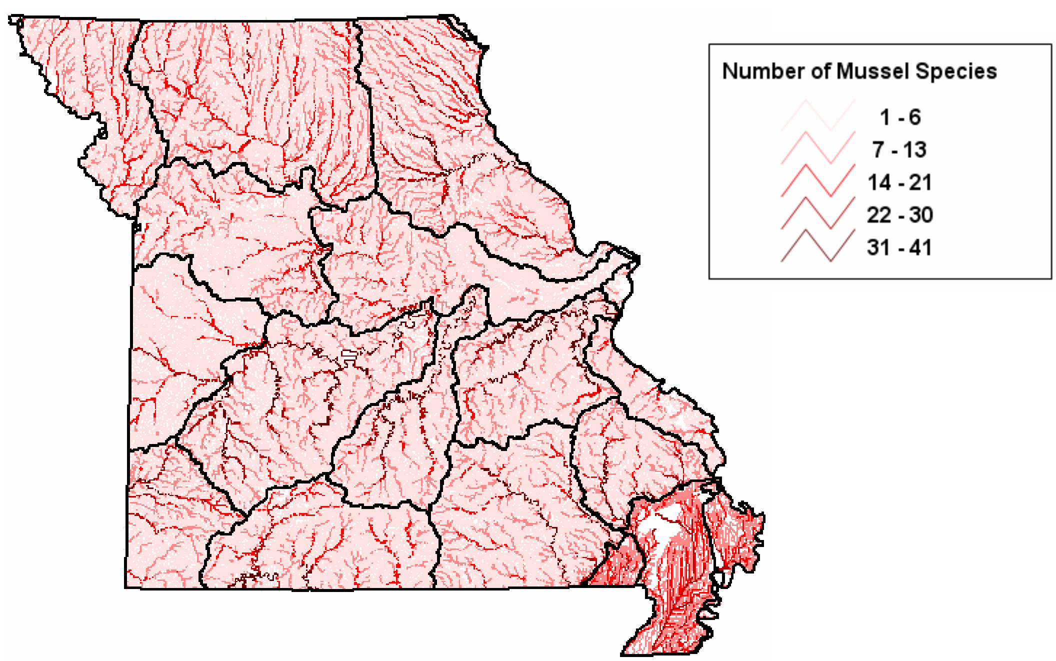


Figure 4.12. Map showing predicted mussel species richness by stream segment.

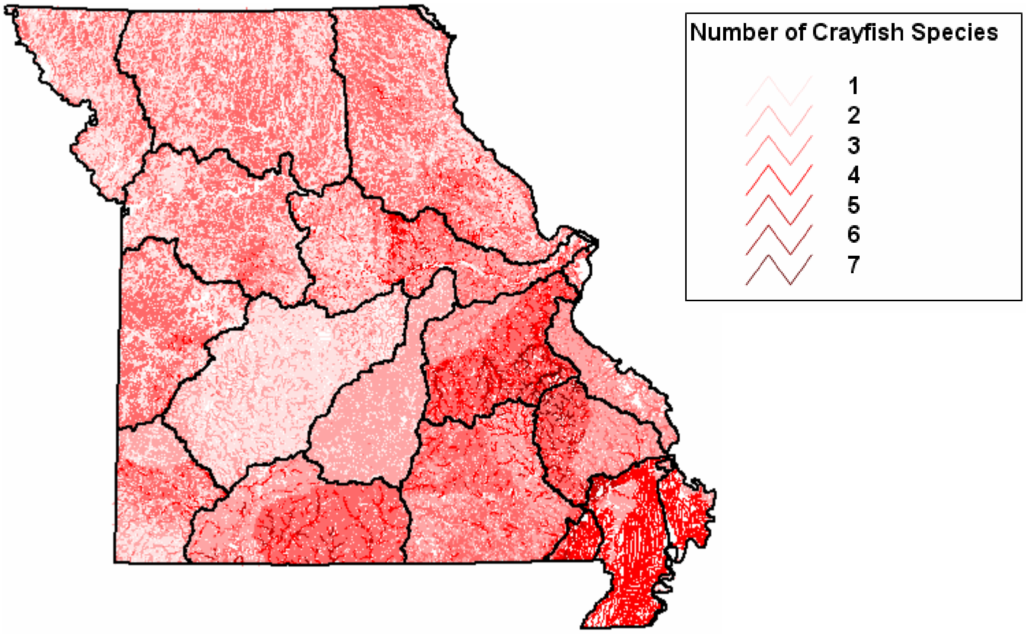


Figure 4.13. Map showing predicted crayfish species richness by stream segment.

Highest richness (140-146 species), across all taxonomic groups, occurs within the Ozark Aquatic Subregion, and more specifically within the lower Meramec River, just before it empties into the Mississippi River near St. Louis. This same stretch of stream contains the highest richness values for both fish (100-102 species) and mussels (40-41 species). Highest richness values for crayfish again occurred in the Ozark Aquatic Subregion, but the highest values (7 species) occurred within the lower Big River, which is a tributary to the Meramec River and the upper St. Francis River.

Within the Central Plains Aquatic Subregion (CP), highest overall (98-100 species) and mussel (35-37 species) species richness occurs at the outlet of the Salt River just before it empties into the Mississippi River. Highest fish species richness (67 species), within the CP occurs within the Osage River just before it enters the Ozark Aquatic Subregion. Highest crayfish species richness (4 species) occurs in large number of smaller streams across a broad region along the boundary between the CP and the Ozarks.

Within the Mississippi Alluvial Basin Aquatic Subregion (MAB), highest overall (102 species) and mussel (28 species) species richness occurs within the lower St. Francis River, which forms the boundary between Missouri and Arkansas. Highest fish species richness (73 species), within the MAB occurs within the Black River just below where this rivers exists the Ozark Aquatic Subregion. Highest crayfish species richness (6 species) occurs within the Black River near the Missouri/Arkansas border and in the Castor River near the boundary between the Ozarks and the MAB.

As expected, and as Figures 4.10 – 4.13 show, our models revealed a strong association between species richness and stream size. On average, across the entire state, headwaters were predicted to contain 20 species (Standard Deviation (SD) = 8), creeks 36 species (SD=12), small rivers 62 (SD=18), and large rivers 93 (SD=27). The fact that the standard deviation of species richness also increases with stream size is both a reflection of the higher number of species and likely also the greater statewide variation in local community composition for these larger streams. Appendix 4.2 provides maps of species richness for each individual stream size class across all taxonomic and for each individual taxonomic groups. These maps show that highest species richness for small and large rivers occurs within the Ozark Aquatic Subregion, while highest species richness for headwaters and creeks actually occurs within the MAB.

Richness patterns for species of special concern (globally listed, G1-G3 species) generally follow the patterns of overall species richness. The highest concentrations of globally rare, threatened and endangered species occur within the lower mainstems of the largest Ozark Rivers (Figure 4.14). As the gap analysis chapter will show, this represents a tremendous challenge for resource managers because of the extremely large land area that must be managed in order to protect or restore the ecological integrity of these large rivers and also the limited amount of public land holdings along these rivers and within their watersheds.

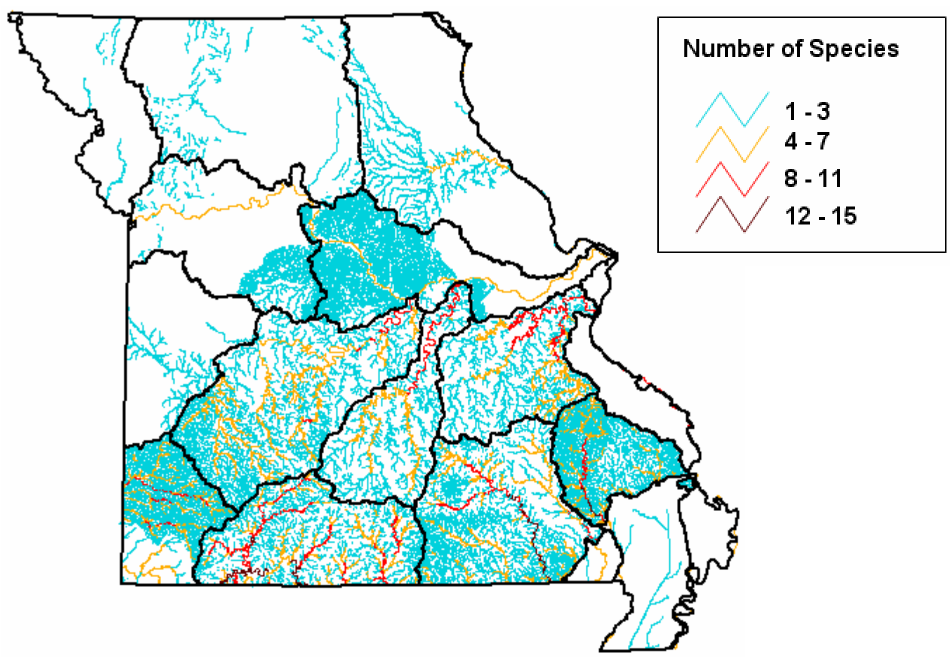


Figure 4.14. Map showing predicted richness of all globally rare, threatened and endangered fish, mussel and crayfish species (G1-G3 species) by stream segment.

4.6 Model Accuracy

Assessing the accuracy of predicted distribution models is a difficult endeavor. Model inaccuracy or error can be either actual or apparent (Schaefer and Krohn 2002) and can result from various factors. For empirically developed models, actual error can result from a variety of sources, including; a) an insufficient number of collection records, b) using inappropriate or omitting critical predictor variables, c) use of proximate predictor variables, d) dataset is unrepresentative of the full range of variation in the predictor variables, e) statistical limitations in the modeling approach used, f) inappropriate modeling procedures, g) errors in the data used to build the model, or h) human error during the course of constructing the model (Fielding 2002; Scott et al. 2002). Sources of apparent error are difficult to quantify and separate from actual errors, yet in many instances may significantly inflate the errors of a model (Boone and Krohn 2002). Most sources of apparent error result from inadequate sampling, which can affect either the input datasets used to construct the models or the independent datasets used to test models. The relative influence of inadequate sampling on apparent error is species dependent. It is also dependent upon how the spatial and temporal specificity of the model (e.g., 5 to 10 km of stream and no temporal restrictions) relates to the spatial and temporal specificity of the sampling data (e.g., collections taken over 100 meters of stream during summer).

Types of Error

When predicting presence versus absence, there are two types of errors; commission and omission. Commission errors occur when a model predicts a species to be present in a specific spatial unit when in fact it does not. Omission errors occur when a species is predicted to be absent in a specific spatial unit when in fact it is present. Both of these errors can be further partitioned into actual and apparent error. Generally this further subdivision has been restricted to commission errors (See Boone and Krohn 2002) since inadequate sampling generates uncertainties as to whether or not the species was actually present at the time of the sample, but for various reasons was not detected. If the species was in fact absent then the error in the model is termed actual, however, if the species was present but not detected then the error is termed apparent. Despite the focus on apparent error associated with errors of commission, we believe that apparent error can also comprise a significant portion of omission errors, especially in riverine ecosystems. An apparent omission error occurs when your model does not predict a species to occur in a given spatial unit when in fact it does, but the occurrence of that species does not meet the objective of your model. For instance, if your objective were to predict stream segments where a given species has a high probability of maintaining a population, then those species that occur in the segment but are unable to maintain a viable population would actually represent apparent errors. Such “stray” occurrences are likely quite common for wide-ranging species with good dispersal capabilities. Also, with human modifications to the landscape, like headwater impoundments that are stocked with various combinations of game species, it is not unusual to find largemouth bass, bluegill, crappie, or catfish in midwestern streams where these species historically did not occur and even today only occur as the result of continuous escapements from these man-made waterbodies (Pflieger 1997).

Independent Testing Data

There are many ways to test model accuracy (See Hand 1997 or Fielding 2002 for a review). The most unbiased measure of accuracy is to test your model on data that are completely independent of those used to generate the model (Fausch et al. 1988; Fielding 2002). Following this advice we obtained several independent datasets to test the accuracy of our predictive models. We obtained 80 fish collections from the Missouri Department of Conservation (MDC) that were collected as part of the Missouri Resource Assessment and Monitoring Program during 2001 and 2002. A total of 147 crayfish collections were obtained from the MDC and the Missouri Cooperative Fish and Wildlife Research Unit. These data were collected during the years of 1991-2002 for various research projects. Finally, we received data for 51 mussel collections from the MDC that were collected during 2000-2002 as part of a long-term statewide inventory and monitoring effort. Table 4.3 provides the percent commission, omission, and overall accuracy statistics for each of the taxonomic groups.

Table 4.3. Percent commission, omission, and overall accuracy statistics for each taxonomic group.

Taxa	Overall	Commission	Omission
Crayfish	48	52	11
Fish	51	48	10
Mussel	36	64	6
Average	45	55	9

4.7 Discussion and Limitations

All species range maps are predictions about the occurrence of those species within a particular area (Csuti and Crist 1998). The purpose of our species distribution maps is to provide more precise information about the current distribution of individual native and nonnative species within their general ranges. With this information, better estimates can be made about the actual amounts of habitat area and the nature of its configuration.

Our species distribution maps were produced at 1:100,000 and are intended for applications at the landscape or *beta* scale (homogeneous areas generally covering 1,000 to 1,000,000 ha and stream segments ranging from 10 to 100 km, which are made up of multiple local biotic communities). Applications of these data to local site-level analyses are likely to be compromised by finer-grained patterns of environmental heterogeneity not captured within our models. The models presented in this report should be viewed as testable hypotheses as their suitability will vary with each given application.

Gap analysis uses the predicted distributions to evaluate the conservation status native species relative to existing land management (Scott et al.1993). However, the resulting distribution maps can be used to answer a wide variety of management, planning, and research questions relating to individual species or groups of species. In addition to the maps, great utility may be found in the specimen collection records comparisons and literature that are assembled into databases used to produce the maps.

Range maps were once viewed as being mainly of interest to naturalists, taxonomists, and biogeographers. However, as resource agencies shift their emphasis from species- and site-specific management to conserving biodiversity over large regions it is becoming increasingly apparent that having precise and accurate range maps is critical to effective conservation. By using GIS and a watershed-based approach to generate range maps for freshwater biota in Missouri we were able to overcome the limited accuracy, precision and utility of hardcopy range maps found in field guides or taxonomic texts. Our GIS-based range maps and associated relational databases allow users to easily generate and visually display a variety of important biological statistics for 315 species to assist with planning, management, and research at several spatial scales. The electronic format of these databases also permits easy editing and updating of distributional data and sharing information over the Internet.

While developing our GIS-based range maps we came to two important realizations. The first realization is that, at present, spatially integrating biological survey data among individuals or agencies is a difficult task, to put it mildly. However, this need not be the case and it is our hope that some day sharing biological data among individuals or agencies will be a relatively “painless” and common practice. For this to happen federal, state and tribal resource agencies and university researchers must recognize the benefits of using globally standardized species codes like those provided by ITIS and spatially linking their collection data to nationally standardized geospatial databases like the NHD and HU coverage. This recognition must be accompanied by administrative directives or even agency-wide policies, which encourage these practices by those responsible for collecting or managing biological survey data. Only when these most basic challenges have been overcome can we then begin to address the equally important challenges to sharing biological data outlined by McLaughlin et al. (2001) and Bonar and Hubert (2002).

The second important realization is that, when it comes to the freshwater resources of our nation, we are by no means beyond the age of exploration. There needs to be a rekindled interest in the intense and geographically extensive biological surveys that were once so prevalent in the late 1800’s and early 1900’s due largely to the emphasis placed upon such activities by the U.S. Commission on Fish and Fisheries and several newly formed state fish and game agencies (Hubbs 1964). Our databases show that even in a relatively data “rich” state like Missouri, only 0.03% of the total stream miles have been sampled for fish, mussels, and crayfish. Also, many watersheds have never been sampled for any taxonomic group and a surprising number of watersheds only have less than three samples. Without field data we must resort to modeling or sheer speculation to generate any sort of understanding about the freshwater biota inhabiting these watersheds. Such speculations are especially problematic for conservation efforts directed at rare, threatened or endangered species. Fortunately we now have the ability in Missouri to identify these information gaps and more importantly we can use our databases to develop optimized sampling strategies for filling these gaps.

Habitat-affinity data are lacking for many species, especially mussels and crayfish. There is an obvious need for more basic life-history research. Since habitat affinities often change with life stage there is also a need for life-stage specific habitat-affinity research. Also, most habitat-affinity information that is available pertains to local habitat factors such as depth, velocity and substrate. This “microhabitat” information cannot be used within a GIS to predict a distribution of a species throughout the watersheds in which they are known to occur unless we can first accurately map or model depths, velocities and substrates throughout entire watersheds, which is unlikely. What is needed is habitat-affinity information at the meso and macro scales which reveal associations between a species presence and factors such temperature, stream size, gradient, geology, permanence of flow, and special lotic environments such as springs and wetlands.

Our predictive models utilized local explanatory variables. We firmly believe that our models could be substantially improved by incorporating watershed variables as predictors as well as by getting more detailed temperature data for valley segments. Through a grant from the Missouri Department of Conservation, MoRAP has recently begun developing these very data for every reach of stream within the 1:100,000 NHD. Once completed the models for all 315 species should be reconstructed using this broader suite of potential predictor variables.

The accuracy statistics of our predictive models are very misleading. There are many problems associated with this accuracy assessment related to spatial and temporal sampling “inadequacies” of the independent datasets and with the inherent difference in what we are trying to predict (i.e., biological potential) versus the fact that most of the stream segments sampled in these independent datasets were degraded to some degree. In fact, some of the sites are highly degraded and in such instances we would expect very little correspondence between our predicted assemblage and the assemblage that presently occupies the site. A proper evaluation of the accuracy of our models will require a separate project that identifies relatively high quality sites, which are then sampled intensively throughout relatively long stretches of stream during several seasons and over a period of several years.

CHAPTER 5

Developing Local, Watershed, and Upstream Riparian Management Status Statistics for Each Stream Segment

For fish it is necessary that a considerable stretch of territory, or even an entire stream, be set aside; and this adds to the expense and difficulties of securing and controlling the area.

Henry B. Ward, 1912
42nd Meeting of the American Fisheries Society

5.1 Purpose

- Assess representation of biotic and abiotic elements of biodiversity within the existing matrix of public lands.
- Assist with conservation planning by providing decision makers with information on which to base the selection of new conservation areas and the expansion or change in management of existing conservation areas.

5.2 Introduction

To fulfill the analytical mission of GAP, it is necessary to assess the representation of mapped elements of biodiversity within existing public land holdings and management status categories. As will be explained in the gap analysis section, these assessments do not measure viability, but are a start to assessing existing, and the likelihood of future, threats to a biotic element through habitat conversion, the primary cause of biodiversity decline. We use the term “stewardship” in place of “ownership” in recognition that legal ownership does not necessarily equate to the entity charged with management of the resource, and that the mix of ownership and managing entities is a complex and rapidly changing condition not suitably mapped by GAP. We emphasize, however, that GAP only identifies private land as a homogeneous management status category and does not differentiate individual tracts or owners, unless the information was provided voluntarily to recognize a long-term commitment to biodiversity maintenance.

The purpose of assessing the management status of biotic and abiotic elements of biodiversity is to identify the possible need for changes in management status for the distribution of individual elements or areas containing distinct species, communities, or ecosystems or areas of high ecological diversity. While it will eventually be desirable to identify specific management practices for each tract of public land, and whether they are beneficial or harmful to each biodiversity element, GAP currently uses a scale of 1 to 4 to characterize the relative degree of maintenance of biodiversity for each tract. A status of “1” denotes the highest, most permanent level of maintenance, and “4” represents the lowest level of biodiversity management, or unknown status. This is a highly subjective area, and we recognize a variety of limitations in our approach, although we maintain certain principles in assigning the status level. Our first principle is that land ownership is not the primary determinant in assigning status. The second principle is that while data are imperfect, and all land is subject to changes in ownership and management, we can use the intent of a land steward as evidenced by legal and institutional factors to assign status. In other words, if a land steward institutes a program backed by legal and institutional arrangements that are intended for permanent biodiversity maintenance, we use that as the guide for assigning management status.

The characteristics used to determine management status are as follows:

- Permanence of protection from conversion of natural land cover to unnatural (human induced barren, exotic-dominated, arrested succession).
- Relative amount of the tract managed for natural cover.
- Inclusiveness of the management, i.e., single feature or species versus all biota.
- Type of management and degree that it is mandated through legal and institutional arrangements.

The four management status categories are generally defined as follows (after Scott et al. 1993, Edwards et al. 1995, Crist et al. 1995):

Status 1: An area having permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a natural state within which disturbance events (of natural type, frequency, and intensity) are allowed to proceed without interference, or are mimicked through management. (e.g. Research natural areas)

Status 2: An area having permanent protection from conversion of natural land cover and a mandated management plan in operation to maintain a primarily natural state, but which may receive use or management practices that degrade the quality of existing natural communities. (e.g. Wilderness areas)

Status 3: An area having permanent protection from conversion of natural land cover for the majority of the area, but subject to extractive uses of either a broad, low-intensity type or localized, high-intensity type. It also confers protection to federally listed endangered and threatened species throughout the area (e.g., national forests).

Status 4: Lack of irrevocable easement or mandate to prevent conversion of natural habitat types to anthropogenic habitat types. Allows for intensive use throughout the tract. Also includes those tracts for which the existence of such restrictions or sufficient information to establish a higher status is unknown (e.g., private).

5.3 General Methods

The GAP stewardship coverage for Missouri was used in conjunction with the Valley Segment coverage to identify stream segments flowing through public lands. A customized ArcView tool was used to identify those segments that have the majority of their length ($\geq$ 51%) within public lands. These segments were then further attributed with the agency responsible for the management of the surrounding tract of land and also the four GAP management status categories described above. Another Arc Marco Language algorithm was used to calculate the percentage of each stream segment's watershed and upstream drainage network that occurs within each of the four GAP management status categories. Since the watersheds of many of the stream segments within Missouri extend beyond the state boundary, the stewardship coverages for the neighboring states of Arkansas, Iowa, and Kansas were merged with that of Missouri. With these attributes users can now select any of the approximately 154,000 individual stream segments within Missouri and see which segments are flowing through public lands, and what percentage of the overall watershed and upstream drainage network is within public ownership by GAP management status category.

5.4 Source Data

National Hydrography Dataset (NHD)

We used the 1:100,000 stream networks as developed by the Missouri Resource Assessment Partnership (MoRAP) for the Missouri Aquatic Gap Project. This stream network is an altered and enhanced version of the *Initial Release* of the 1:100,000 National Hydrography Dataset (NHD) that was developed by the United States Geological Survey (USGS) and the United States Environmental Protection Agency (USEPA). We acquired the *Initial Release* of the NHD in 1999.

Digital Elevation Model (DEM)

We used the United States Geological Survey (USGS) 30-meter resolution digital elevation models. Individual DEM components for the study area were combined using the ArcInfo Grid command MOSAIC. For storage and processing considerations the DEM was converted to a rounded integer grid. We filled the DEM sinks using the ArcInfo Grid FILL command.

GAP Stewardship Layers

We combined GAP stewardship layers for the following four states into one shapefile in order to accurately generate watershed and upstream riparian statistics for each of the stream segments within Missouri (Figure 5.1). The methods and standards used to create each of these stewardship layers can be found in the references or at the web pages provided below.

1. Arkansas – Arkansas Gap Analysis Stewardship Coverage developed by the University of Arkansas Center for Advanced Spatial Technologies (Smith et al. 1998).
2. Iowa – Iowa Gap Analysis Stewardship Coverage developed by the Iowa Cooperative Fish and Wildlife Research Unit / ISU GIS Facility. The following website provides information on how this coverage was developed for Iowa GAP http://www.iowagap.iastate.edu/

3. Kansas – Kansas Gap Analysis Stewardship Coverage developed by the Kansas Biological Survey, Geographic Information Systems Spatial Analysis Laboratory (GISSAL), Kansas Cooperative Fish and Wildlife Research Unit, Kansas Department of Wildlife and Parks, Kansas Applied Remote Sensing (KARS), University of Kansas, Kansas State University, Kansas GIS Policy Board, U.S. EPA, NASA, U.S.G.S. Biological Resources Division, National Park Service. The following website provides information on how this coverage was developed for Iowa GAP http://www.ksu.edu/kansasgap/

4. Missouri – Missouri Gap Analysis Stewardship Coverage developed by the Missouri Resource Assessment Partnership (MoRAP), University of Missouri (Haithcoat and Drobney 2002).

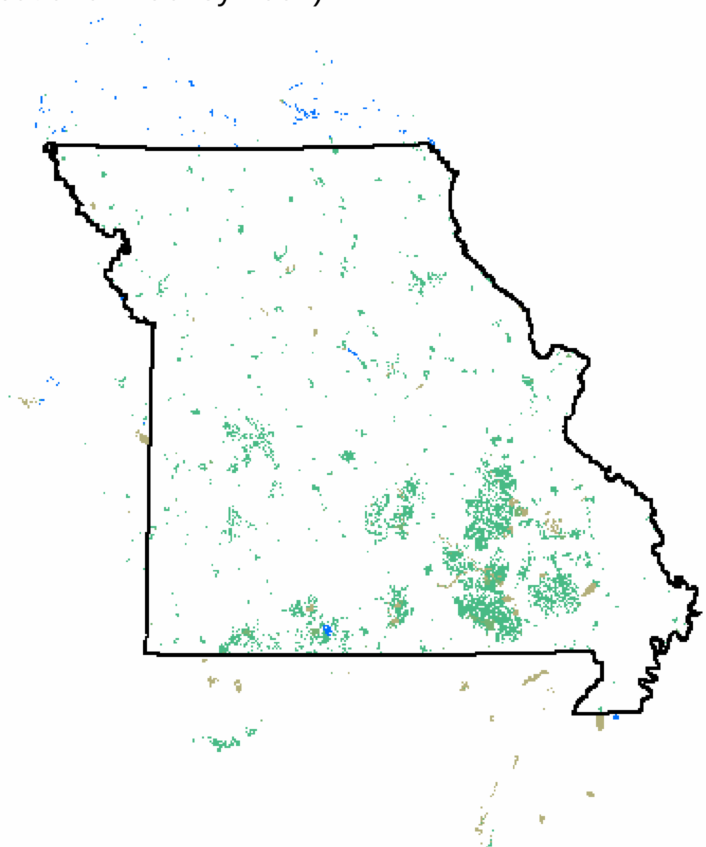


Figure 5.1. Map showing the GAP stewardship coverage for Missouri and portions of Arkansas, Iowa, and Kansas that were collectively used to calculate the percentage of the watershed and upstream drainage network falling within each GAP management status category for each stream segment within Missouri.

5.5 Detailed Methods

In order to get generate accurate management status statistics for all stream segments in Missouri, (excluding the Missouri, Mississippi and Des Moines Rivers) we had to first acquire the GAP stewardship coverages for the states of Arkansas, Iowa, Kansas, and Missouri. Because of slight differences in how these state based coverages were developed, some editing of the stewardship coverages was necessary. During this process we found that there needs to be more stringent standards placed on how public lands are categorized into the four GAP management status categories. When merging the stewardship coverages of adjacent states (Arkansas, Iowa, and Kansas) with Missouri's coverage, we found many discrepancies in how public lands were placed into the four management status categories. This has serious implications for regional assessments of biodiversity protection. Regional committees are likely needed to address this important issue.

We used GAP management categories from one through four as described in section 5.2. In addition, we wanted to exclude "public land" that was inundated by large lakes. This was accomplished by excluding waterbodies from the stewardship layers. The most common landowner for the large reservoirs was the US Army Corps of Engineers. The primary purpose behind this process was to keep the stream reaches and those portions of their drainages that are inundated by large manmade impoundments out of the public land stewardship computations (Figure 5.2).

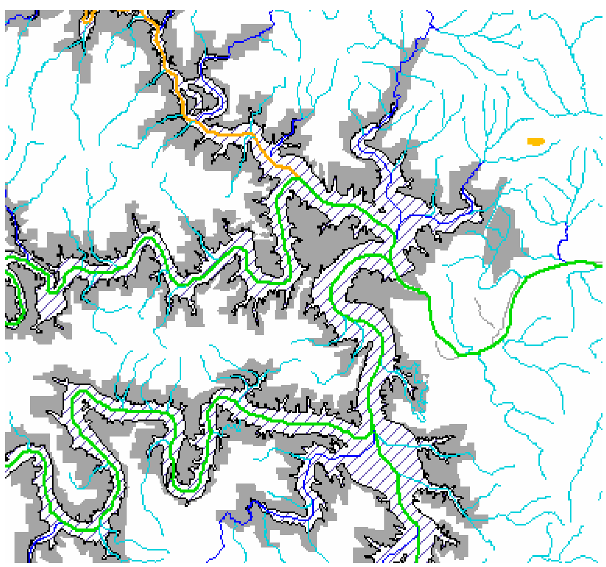


Figure 5.2. An example of how public property falling within large manmade impoundments was removed from all management status calculations. The solid grey area falling outside of the high pool line was included in all management status calculations (local, watershed, and upstream riparian) while the inundated portions (grey hatching) were excluded from all such calculations.

Local Stewardship and Management Status Statistics

The GAP stewardship coverage for Missouri was used in conjunction with our 1:100,000 Valley Segment coverage to identify individual stream segments flowing through public lands. A customized ArcView tool, developed by The Nature Conservancy's Freshwater Initative, was used to identify and attribute (binary 1 or 0) those segments that have the majority of their length (≥ 51%) within public lands. These segments were then further attributed with the agency responsible for the management of the surrounding tract of land (Figure 5.3) and also the four GAP management status categories described above (Figure 5.4).

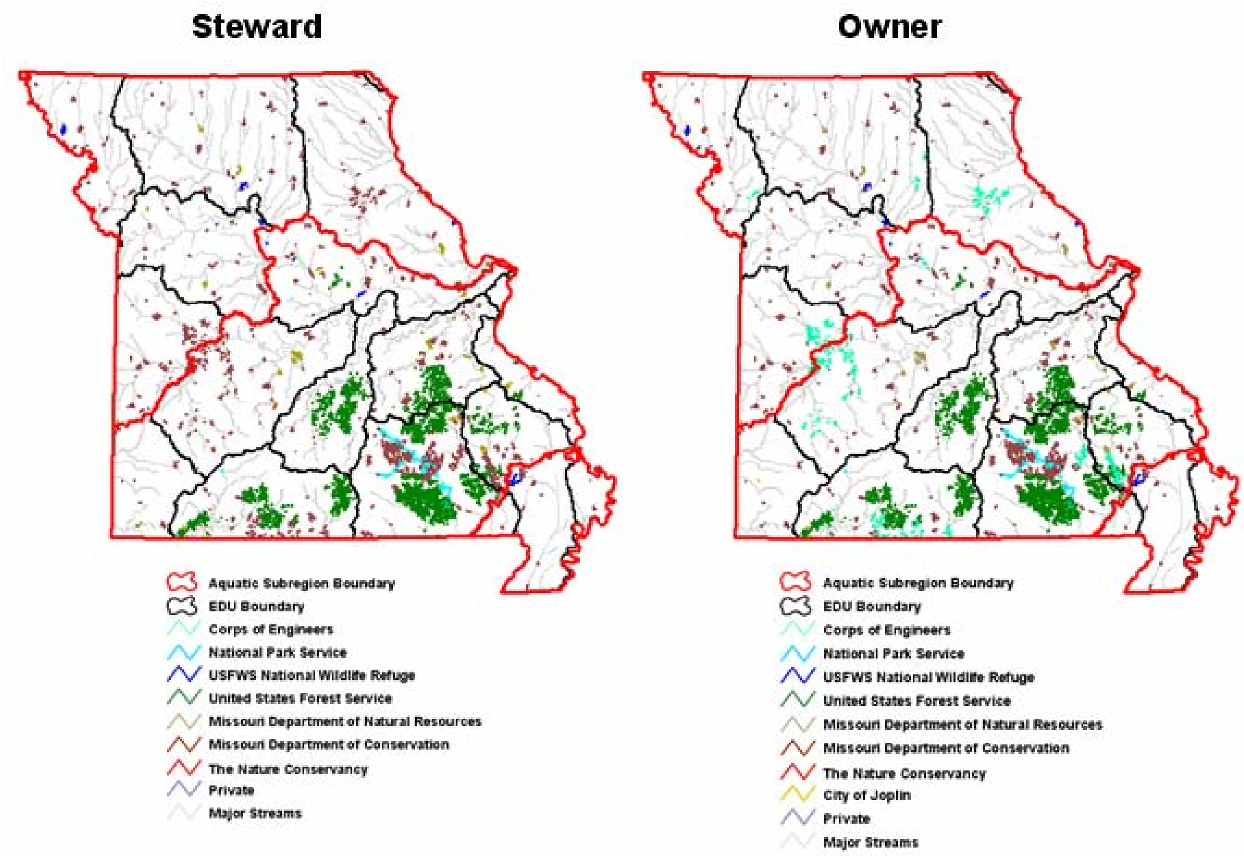


Figure 5.3. Maps of Steward and Owner for stream segments that have the majority of their length within any public land or private lands managed for the protection of biodiversity.

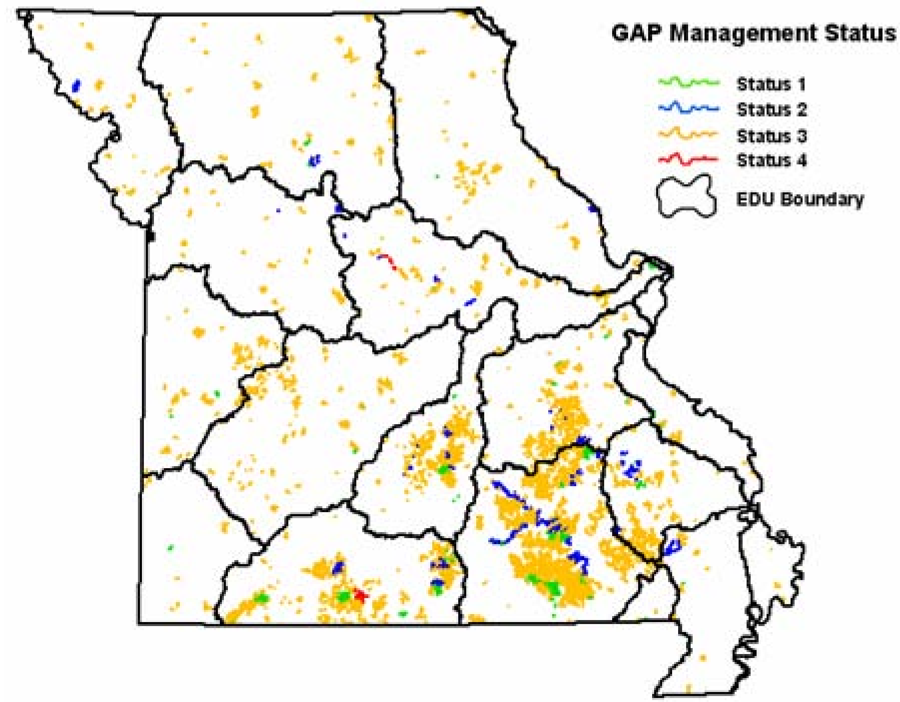


Figure 5.4. Map of stream segments with greater than 50% of their length flowing through public land displayed according to the four GAP management status categories.

Upstream Drainage Network Management Status Statistics

Beyond simply identifying stream segments flowing through public lands, we also computed the percent of each segment's upstream drainage network that is contained in each GAP management status category. After each stream segment was attributed with the appropriate GAP management status, for the land through which it flows, we used the ARC/INFO TRACE ACCUMULATE command to calculate the total length of stream, in each GAP management status, within the entire drainage network above each stream segment. These length computations were then converted to a percent of the upstream network for each segment (Table 5.1 and Figure 5.5).

Table 5.1. An example of the upstream drainage network and overall watershed statistics generated for each stream segment in the Missouri Valley Segment coverage. Table shows, for three individual stream segments, the percent of the upstream network and watershed falling in all public lands (GAP 1-4) and the percent falling in lands classified as GAP management status 1 or 2 (GAP 1-2).

Stream Segment ID	Upstream Network GAP 1-4	Watershed in GAP 1-4	Upstream Network GAP 1-2	Watershed in GAP 1-2
10300101 8377	11.49%	14.48%	0.0%	2.78%
10300101 5579	11.47%	29.61%	0.0%	0.96%
10300101 5888	10.76%	8.44%	10.76%	8.44%

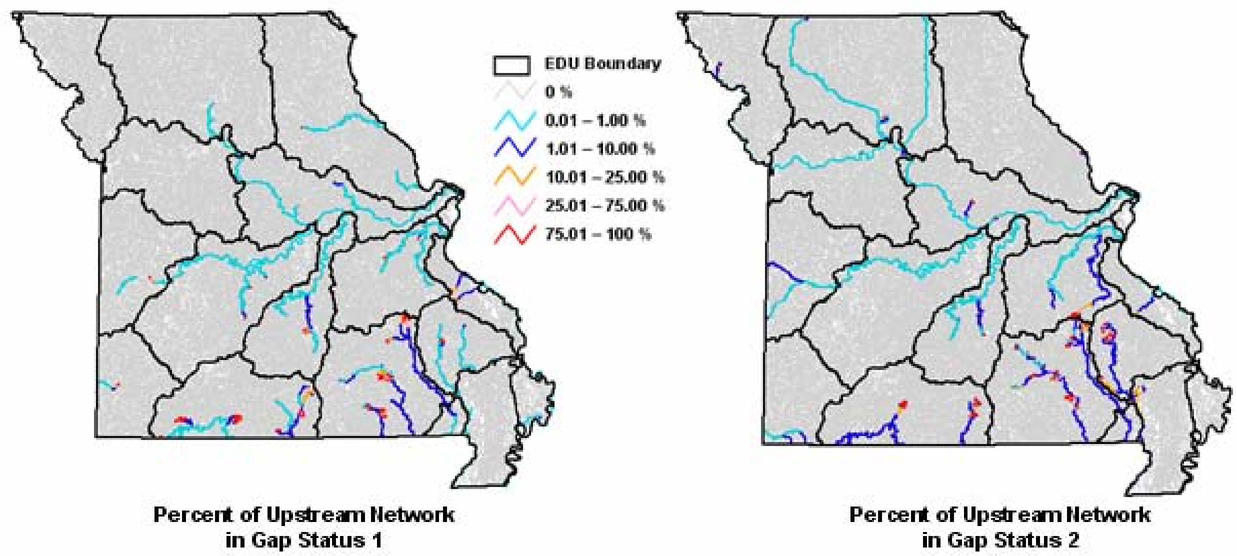


Figure 5.5. Maps showing the percent of the upstream network of each stream segment that is contained within lands classified as GAP management status 1 and 2.

Overall Watershed Management Status Statistics

Since the ecological integrity of rivers is significantly influenced by conditions within the watershed and because much of the public land in Missouri is situated in the uplands, we also took steps to compute the percent of each stream segment's watershed that falls within in each GAP management status category. Beginning with the 1:100,000 Valley Segment coverage, all secondary channels were removed and the resulting network run through an Arc Macro Language (AML) created by The Nature Conservancy's Freshwater Initiative (sheds.aml; TNC 2000). This AML uses the arcs in the stream network in conjunction with a digital elevation model (DEM) to generate "segmentshed" polygons for each individual stream segment in the network (Figure 5.6). A segmentshed represents the immediate land area that drains to a given stream segment. The corresponding stream segment and segmentshed polygon share a unique identifier that allows for efficient attribute transfer between the lines and polygons and vice versa.

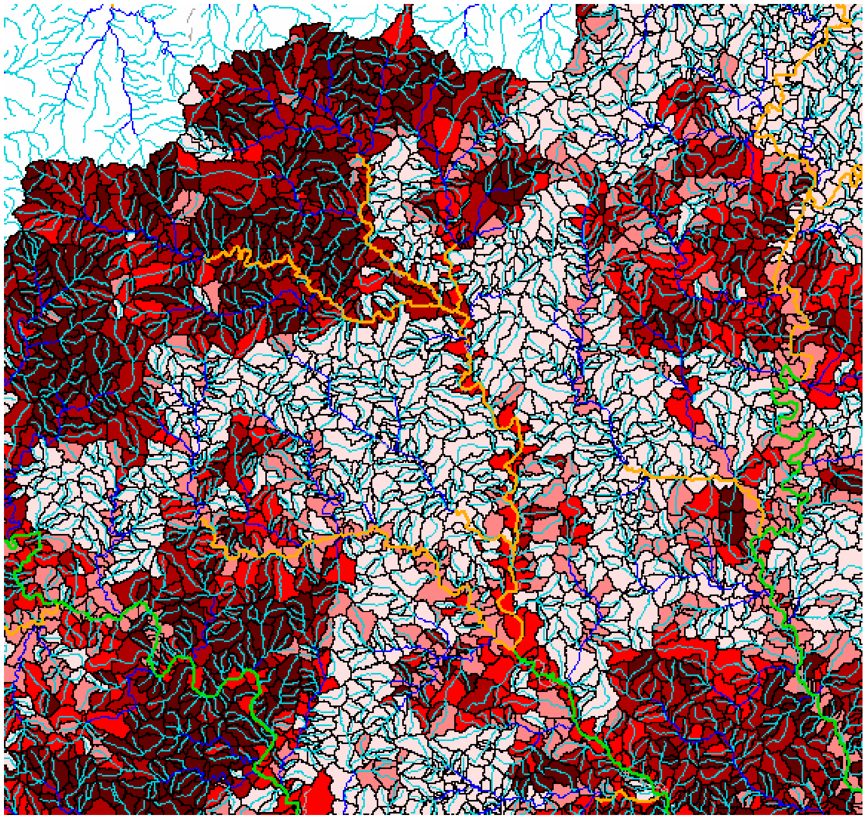


Figure 5.6. An example of how segmentsheds were created for each stream segment in the 1:100,000 Valley Segment coverage in order to calculate the watershed percentages, by GAP management status, for each individual stream segment. In this example the segmentsheds are displayed in a graduated color showing the percent of each polygons watersheds falling in all public lands (GAP 1-4). Darker colors indicate a higher percent of public land within the watershed. Corresponding streams are colored by stream size; headwater-light blue; creek-dark blue; small river-orange; large river-green.

We used the polygonal segmentshed coverage in conjunction with the multistate GAP stewardship coverage to find the area of each polygon that was contained in each management status category. Data was loaded into ArcView and the area of each management status category was calculated for each individual segmentshed polygon (see Figure 5.6). These data were then transferred from the segmentshed polygon coverage to the corresponding stream network coverage using a common identifier. The TRACE ACCUMULATE command was then used to summarize the area of each management status category within the entire watershed of each stream segment. These area computations were then converted to a percent of the watershed (see Table 5.1 and Figure 5.7).

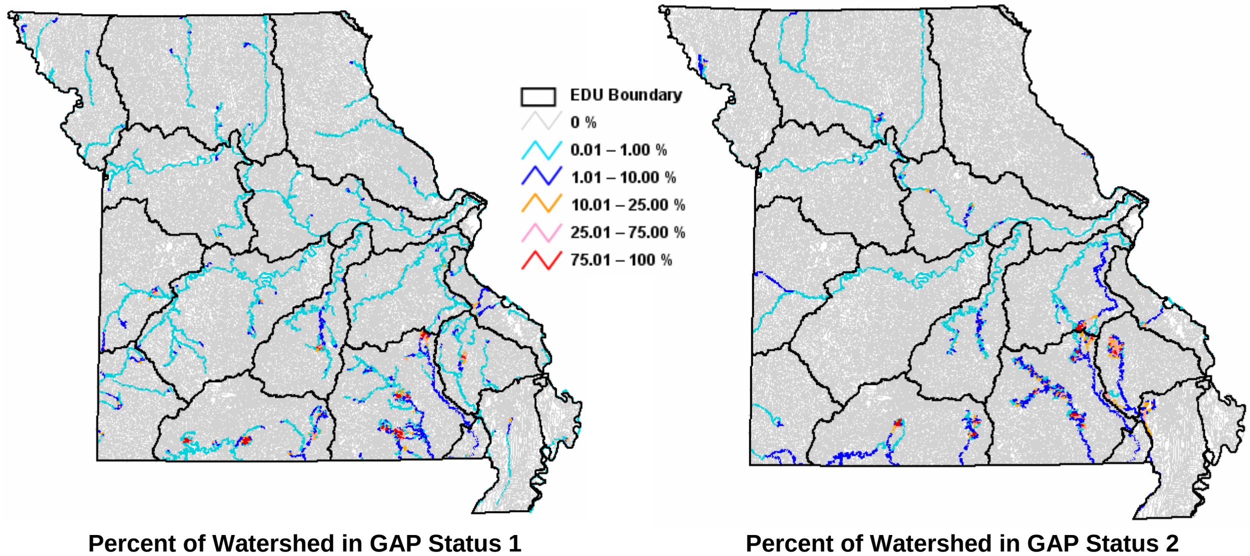


Figure 5.7. Maps showing the percent of the watershed of each stream segment that is contained within lands classified as GAP management status 1 and 2.

5.6 Results

Local Stewardship and Management Status

Just over 5% (9,373 km) of the 174,063 total kilometers of stream within Missouri are contained within the existing matrix of public lands (Table 5.2 and 5.3). The vast majority of these streams (85.2%) flow through lands classified as GAP management status 3. Less than 1% (0.8%) flow through lands that are primarily managed for the long term maintenance of biodiversity (i.e., management status 1 or 2) (Table 5.3). As expected, the number of kilometers within public ownership generally decreases with stream size, however, the relative percentages for each class are fairly similar (Table 5.4). Large rivers have the highest relative percentage within management status 1 or 2 (153 km, 4.3%). Most of these 153 km occur within the Ozark National Scenic Riverways (ONSR) and the Eleven Point National Wild and Scenic River. The designated boundary for the OSNR contains 216 km of the Current and Jacks Fork Rivers, which is owned and managed by the National Park Service, and was the first National Scenic Riverway to be established by Congress on August 24, 1964. However, not all of the 216 km within this boundary is presently owned by the NPS or falls within management status 1 or 2. Land within the Eleven Point National Wild and Scenic River boundary is owned and managed by the U.S. Forest Service and was one of the original eight wild and scenic river units established by Congress in 1968. Approximately one half of the total length of stream within the designated 71 km section of the Eleven Point River is privately owned.

Table 5.2. Number of kilometers and percent of total kilometers within public land by GAP management status. Note: Does not include statistics for MO and MS Rivers.

GAP Management Status	Kilometers	Percent of Total Within Public
1	546	5.8
2	795	8.5
3	7,990	85.2
4	47	0.5
All Public Lands	9,377	100

Table 5.3. Number of kilometers and relative percentage statistics for stream segments flowing through any public land (All Public Lands) and those flowing through land classified as GAP management status 1 or 2, broken down according to stream size classes. Note: Great Rivers (MO and MS Rivers) were not included in the assessment and the relative percentage statistics for "All Sizes" exclude the total kilometers (1,664) for this stream size class.

Stream Size	Total Km	All Public Lands	Percent in All Public Lands	Km in Status 1 or 2	Percent in Status 1 or 2
Headwater	129,394	7,314	5.65	776	0.60
Creek	27,624	1,075	3.89	195	0.71
Small River	11,904	699	5.87	214	1.80
Large River	3,547	288	8.13	153	4.31
Great River	1,665	NA	NA	NA	NA
All Sizes	174,134	9,377	**5.4**	1,338	**0.8**

Table 5.4. Number of kilometers and relative percentage statistics for stream segments flowing through each GAP management status, broken down according to stream size classes. Note: Great Rivers (MO and MS Rivers) were not included in the assessment and the relative percentage statistics for "All Sizes" exclude the total kilometers (1,664) for this stream size class.

Stream Size	Total Km	GAP1 Km	Percent in GAP1	GAP2 Km	Percent in GAP2	GAP3 Km	Percent in GAP3	GAP4 Km	Percent in GAP4
Headwater	129,394	374	0.29	403	0.31	6,493	5.02	45	0.04
Creek	27,624	85	0.31	109	0.40	881	3.19	0	0.00
Small River	11,904	40	0.34	175	1.47	483	4.06	0	0.00
Large River	3,547	47	1.32	108	3.04	134	3.77	0	0.00
Great River	1,665	NA	NA	NA	NA	NA	NA	NA	NA
All Sizes	174,134	546	0.3	795	0.5	7,990	4.6	45	0.03

With nearly 5,000 km of stream, the U.S. Forest Service (Mark Twain National Forest) has, by far, the largest public holding of streams in Missouri (Table 5.5). Nearly 53% of all the streams flowing through public lands in Missouri are managed by the U.S. Forest Service. The vast majority of these streams are classified as headwater streams. The US Forest Service also has nearly 718 km of stream that flow through lands classified as management status 1 or 2, which is about 2 times higher than the National Park Service and at least 5 times higher than any other management agency (Table 5.5). The next largest public holding of streams in Missouri is contained within lands owned and managed by the Missouri Department of Conservation (MDC) (2,527 km) followed by the US Army Corps of Engineers (COE) (962 km) and the Missouri Department of Natural Resources (381 km). However, most lands owned by the COE are managed by

MDC. Collectively, Figure 5.4 and Table 5.5 illustrate the complexity of public stewardship of the stream resources in Missouri. When you consider the interconnectedness of stream networks, and thus the spatial interdependence of stream conditions, it is quite apparent that the management of streams within the public lands of Missouri is considerably complex, from a logistical and sociopolitical standpoint.

Table 5.5. Statistics by GAP management status for each land steward with lands managed for the protection of biodiversity.

Manager	GAP1 Km	GAP2 Km	GAP3 Km	GAP4 Km	Total Kilometers	% of Total Public
United States Forest Service	380.8	337.4	4,183.8	36.6	4,938.6	52.69
Missouri Department of Conservation	47.0	0.0	3,415.4	0.0	3,462.4	36.94
Corps of Engineers	0.0	0.0	0.0	9.0	9.0	0.10
Department of Natural Resources	23.1	0.0	357.1	0.0	380.2	4.06
National Park Service	43.1	320.0	30.5	0.0	393.6	4.20
USFWS National Wildlife Refuge	0.0	136.7	0.0	0.0	136.7	1.46
The Nature Conservancy	33.7	0.0	0.0	0.0	33.7	0.36
Private	18.1	0.0	0.0	0.0	18.1	0.19

Table 5.6. Statistics by GAP management status for each land owner with lands managed for the protection of biodiversity.

Owner	GAP1 Km	GAP2 Km	GAP3 Km	GAP4 Km	Total Kilometers	% of Total Public
United States Forest Service	384.4	337.4	4,183.8	36.6	4,942.2	52.73
Missouri Department of Conservation	68.9	0.0	2,458.4	0.0	2,527.3	26.97
Corps of Engineers	0.0	0.0	952.3	9.0	961.3	10.26
Department of Natural Resources	24.6	0.0	357.1	0.0	381.7	4.07
National Park Service	19.6	309.7	29.3	0.0	358.6	3.83
USFWS National Wildlife Refuge	0.0	136.7	0.0	0.0	136.7	1.46
The Nature Conservancy	27.5	0.0	4.8	0.0	32.3	0.35
Private	18.1	10.3	1.2	0.0	29.6	0.32
City of Joplin	2.7	0.0	0.0	0.0	2.7	0.03

Most of the public lands within Missouri, and hence most of the streams flowing within public lands, occur within the relatively rugged and agriculturally unproductive Ozark Aquatic Subregion (see Figure 5.4; Table 5.6). Over 8% of the total length of stream within the Ozarks are contained within existing public lands, compared with less than 2% for each of the other two Aquatic Subregions. This pattern of public ownership is consistent with the findings of Scott et al. 2001 who found that most public lands in the United States are situated at higher elevations and in areas of low soil productivity. Differences among the three Aquatic Subregions are particularly evident when all public lands are considered, yet even when the comparisons are restricted to management status 1 or 2 lands, the Ozarks still contains a disproportional amount of the total. The most dramatic differences among the three Subregions are reflected in the statistics pertaining to the percentage of the larger stream size classes flowing within management status 1 or 2 lands. Over 3.5 and 7.0% of the small and large river kilometers within the Ozarks are contained within management status 1 or 2 lands, respectively. In contrast, less than 1% of the streams classified as small river and none of the large rivers, within the Central Plains and the Mississippi Alluvial Basin, are contained in management status 1 or 2 lands. Lands adjacent to these larger streams, within these two Subregions, represent some of the most productive agricultural lands in the state. Consequently, it is not surprising to find limited public land holdings along the major streams within these two Subregions. While the lands adjacent to the larger streams in the Ozarks are also somewhat productive and desirable for development, the aesthetic qualities and recreational value of these larger Ozark streams has permitted and fostered the establishment of nature reserves along at least a few of these streams.

Table 5.7. Stewardship statistics for streams within each of the three Aquatic Subregions of Missouri.

Aquatic Subregion	Stream Size Class	Total Km	Km in Public	Percent in Public	Km in Status 1 or 2	Percent in Status 1or 2
Central Plains	Headwater	49,387	668	1.35	34	0.07
	Creek	13,392	299	2.24	27	0.2
	Small River	6,351	256	4.03	35	0.56
	Large River	1,113	47	4.2	0	0
	All Sizes	70,243	1,270	**1.8**	97	**0.1**
Ozarks	Headwater	73,585	6,543	8.89	716	0.97
	Creek	12,482	731	5.86	140	1.12
	Small River	4,988	427	8.55	175	3.52
	Large River	2,175	235	10.81	153	7.03
	All Sizes	93,230	7,936	**8.5**	1,185	**1.3**
MS Alluvial Basin	Headwater	6,421	105	1.63	27	0.43
	Creek	1,752	47	2.67	27	1.56
	Small River	565	16	2.85	5	0.85
	Large River	192	6	3.36	0	0
	All Sizes	8,929	174	**1.9**	60	**0.7**

Upstream Network and Watershed Stewardship and Management Status

Public ownership of a stream segment does not ensure long-term protection, since everything that occurs within the watershed influences the ecological integrity of that segment (Hynes 1975). This is why we calculated the percent of the watershed and upstream drainage network within public ownership for each stream segment. However, it is difficult to effectively use these data for assessing conservation gaps, because there is a lack of empirical data addressing the question of, “How much is enough?” Is 25, 50, or 75% public ownership within the watershed sufficient to ensure long-term protection? Such thresholds must be identified in a variety of regional settings. As will be discussed in Chapter 8, these data are very useful, however, for conservation planning purposes since they allow you to identify privately-owned stream segments that have a relatively high percentage of their watershed or upstream network in public lands. Such segments can be targeted for future acquisition or the application private land management incentive programs.

As Figure 5.8 illustrates, there are very few larger streams that have a majority of their watershed or upstream network within existing public lands. This represents a real problem for resources managers, especially when you consider that much of our riverine biodiversity and species of special concern are located within these larger streams. It is unlikely that this picture will change in the foreseeable future, and therefore private lands management will be a key to long term maintenance of riverine biodiversity in much of the United States, especially when it comes to larger streams. Figures 5.8 and 5.9 also illustrate the fact that most public lands within Missouri are situated in the uplands, away from the major stream channels. While streams are certainly a reflection of the condition of lands in the uplands, even 90% public ownership of a watershed can have a limited affect on conservation of a particular stream segment if that segment is privately owned and becomes highly altered due to any number of human disturbances that are discussed in the following chapter. This is why our GAP analysis, covered in Chapter 7, focuses on local ownership. We believe that, in terms of reserve design, local public ownership is a necessary first step for the long-term maintenance of riverine biodiversity since even the most ambitious of watershed management efforts (e.g., 90% of watershed in status 1 or 2 lands) can be undone by local human disturbances to the stream segments that are the focus of conservation.

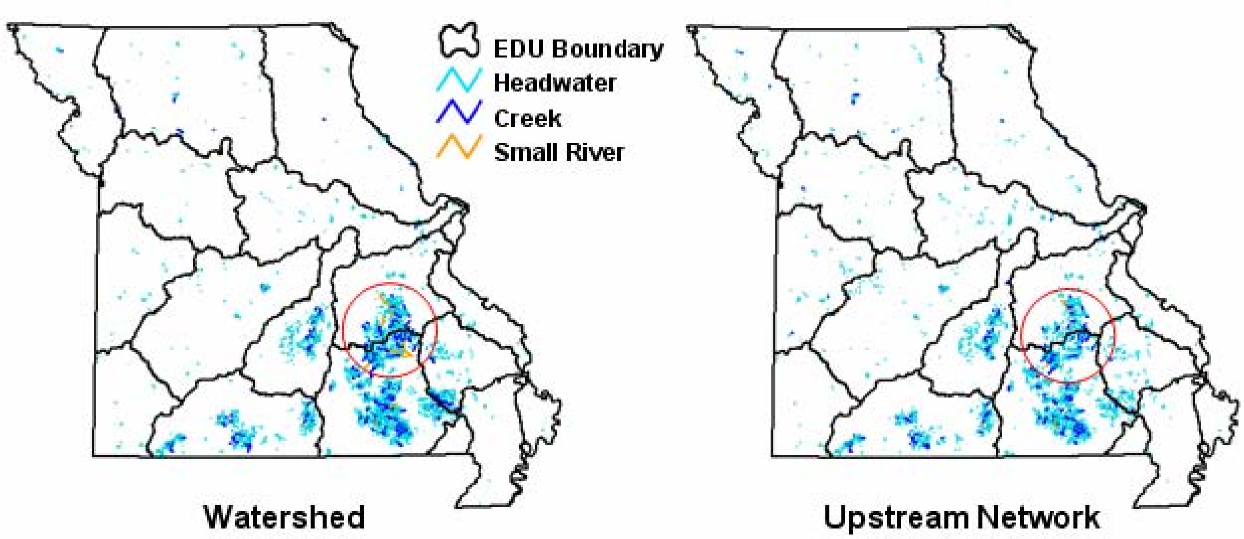


Figure 5.8. Maps of stream segments with greater than 50% of their watershed and upstream drainage network within any public land, broken down by stream size. Note: no stream classified as large river has greater than 50% of its watershed within public ownership. Also note, within the red circle there are fewer streams classified as small river with greater than 50% of their upstream network in public ownership than those having greater than 50% of their watershed in public ownership. This illustrates the fact that most public lands within Missouri are situated in the uplands away from the stream channels.

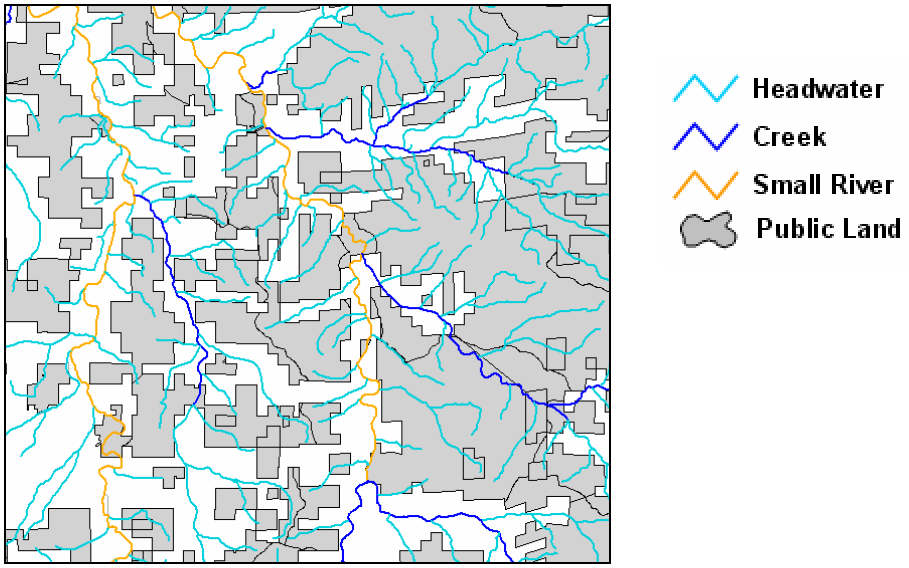


Figure 5.9. Map illustrating the fact that most public lands are situated in the uplands within Missouri.

5.7 Discussion and Limitations

"...careful examination of the conditions prevailing in [nature preserves] will show perfectly distinctly that in many cases the fishes and the aquatic life connected with them are the only elements which are not subject to protection. It is possible to catch fish or to destroy aquatic life,….[yet] you cannot pick the flowers, you cannot break the shrubbery or in any way injure the vegetation of the tract, but if you go fishing it is nobody's business; the fish have to look out for themselves!"

Henry B. Ward, 1912
42nd Meeting of the American Fisheries Society

The above quote by Henry B. Ward is still pertinent today. Most freshwater ecosystems that are contained within status 1 or 2 lands do not receive the same protections as the terrestrial flora and fauna. Fishing, bait collection, and intense recreational use are generally allowed on most of these freshwaters. Consequently, it is reasonable to question the validity/utility of the management status categories as applied to freshwater ecosystems. Certainly, inland waters are a reflection of the conditions of the surrounding land and all lands in the watershed. However, it may be more important to explicitly map fishing/harvest regulations, use restrictions, and the designated uses as administered under the Clean Water Act for each lake, wetland, and reach of stream. This issue must be addressed more fully by future gap analysis projects.

The watershed and percent upstream network stewardship statistics we generated for each stream reach are certainly useful for conservation planning. However, thresholds must be identified so that these data can also be used for gap analyses. Research examining whether 10, 25, 50, or 75% public ownership within the watershed or upstream network is sufficient for long term conservation of ecological integrity must be conducted in various regions across the nation.

East of the Rocky Mountains, it is unlikely we will ever achieve significant public ownership within the watersheds of streams classified as small or large river. As the gap analyses will show in Chapter 7, this represents a significant obstacle for freshwater biodiversity conservation since, if the results for Missouri are any indication, the highest concentration of species of special concern are situated in these larger streams. Considering the limited land holdings in the Midwestern and Eastern United States, private lands management will be critical to the long term conservation of biodiversity within these larger streams.

The land stewardship and management status maps that were used in this report (Missouri, Arkansas, Iowa, and Kansas) represent a compilation of stewardship maps provided by a variety of sources. These maps were created solely for the purpose of conducting the types of gap analyses described in this report and are not suitable for locating boundaries on the ground or determining precise area measurements of individual tracts.

Land ownership and stewardship is constantly changing. The user should also understand that land management status and ownership category may change over time as parcels are bought, sold, and traded. This land stewardship layers used in this report should be considered a “snapshot” of land management status in each of the four states at the time the source data was originally submitted for inclusion into each state Gap Analysis project.

Every effort to reduce the error associated with the combination of spatial data from different sources and native scales was made during the production of the stewardship layers for each state. These stewardship layers, however, were not designed to be used as a legal document or to dispute property boundaries. They were designed to answer questions related to the protection of ecological elements over large areas. Once these areas have been identified, more detailed analyses can be conducted. These data are not a substitute for surveyed data and are not appropriate for analyses at spatial scales greater than 1:100,000.

CHAPTER 6

Developing and Assembling Geospatial Data on Threats and Human Stressors

No adequate vigorous efforts have been made to control [pollution originating outside nature preserves], to correct the errors which have been made, or to keep the waters of such reserves favorable places for the breeding of all aquatic life.

Henry B. Ward, 1912
42nd Meeting of the American Fisheries Society

6.1 Purpose

- Because public ownership does not ensure effective long-term conservation, measures must be taken to account for human stressors that might significantly impair the ecological integrity of those segments currently within public ownership.

- Assist with conservation planning by providing decision makers with quantitative and qualitative information that can be used to identify relatively high quality locations in order to conserve a given conservation target.

- Assist with conservation planning by providing decision makers with quantitative and qualitative information that can be used to identify what factors threaten the ecological integrity of a particular priority location, which can then be used to prioritize management objectives.

- Provide spatially explicit information on human stressors to allow resource managers to pinpoint the specific location of the stressor(s) within the drainage network or watershed.

6.2 Introduction

Land cover serves as means of directly assessing human disturbance to terrestrial ecosystems. Certainly, there are other factors that must be considered (e.g., air pollution and even water pollution), however, the fact that land cover provides a reliable general surrogate for coarse-scale assessments of ecological health of terrestrial ecosystems is likely why none of the projects carried out for the terrestrial component of GAP have explicitly addressed human disturbances. When you consider the multitude of human disturbances affecting riverine ecosystems, the diffuse and cumulative nature of such disturbances, and also the fact that they are often greatly removed from the site of interest, it becomes readily apparent that measures must be taken to account for human disturbances. Failure to do so could lead to misleading statistics in those instances where streams flowing through status 1 or 2 lands are significantly impaired due to hydrologic modification, nonpoint source pollution, or any number of other factors. Certainly, in such instances it would be incorrect to state that these stream segments and their biota are being adequately conserved within the existing matrix of public lands.

6.3 General Methods

Working in consultation with a team of aquatic resource professionals, we generated a list of the principal human activities known to negatively affect the ecological integrity of Missouri streams. We then assembled the best available (i.e., highest resolution and most recent) geospatial data that could be found for each of these stressors. Next, we generated statistics on 65 individual human stressors (e.g., percent urban, lead mine density, degree of fragmentation) for each of the 542 Aquatic Ecological System (AES) polygons in Missouri. We then used correlation analysis to reduce this overall set of metrics into a final set of 11, relatively uncorrelated, measures of human disturbance. Relativized rankings (range 1 to 4) were then developed for each of these 11 metrics. A rank of 1 is indicative of relatively low disturbance for that particular metric, while a rank of 4 indicates a relatively high level of disturbance. The relativized rankings for each of these 11 metrics were then combined into a three number Human Stressor Index (HSI). The first number reflects the highest ranking across all 11 metrics (range 1 to 4). The last two numbers reflect the sum of the 11 metrics (range 11 to 44). This index allows you to evaluate both individual and cumulative effects of the various human stressors. For instance, a value of 418, indicates relatively low cumulative impacts (i.e., last two digits = 18 out of a possible 44), however, the first number is a 4, which indicates that one of the stressors is relatively high and potentially acting as a major human disturbance within that particular ecological unit.

6.4 Detailed Methods

There are a multitude of stressors that negatively affect the ecological integrity of riverine ecosystems (Allan and Flecker 1993; Richter et al. 1997). The first step in any effort to account for anthropogenic stressors is developing a list of candidate causes (U.S. EPA 2000). Working in consultation with a team of aquatic resource professionals, we first generated a list of the principal human activities known to negatively affect the ecological integrity of Missouri streams. We then assembled the best available (i.e., highest resolution and most recent) geospatial data that could be found for each of these stressors (Table 6.1). Fortunately, and somewhat surprisingly, data were available for most stressors. However, for some, such as channelized stream segments, there were no available geospatial data, and efforts to develop a coverage of such segments using a sinuosity index proved ineffective. Most of the geospatial data were acquired from the U.S. EPA, U.S Bureau of Mines, and the Missouri Departments of Conservation and Natural Resources.

Table 6.1. List of the GIS coverages, and their sources, that were obtained or created in order to account for existing and potential future threats to freshwater biodiversity in Missouri.

Data layer	Source
303d Listed Streams	Missouri Department of Natural Resources (MoDNR)
Confined Animal Feeding Operations	MoDNR
Dam Locations	U.S. Army Corps of Engineers (1996)
Drinking Water Supply (DWS) Sites	U.S. Environmental Protection Agency (USEPA)
High Pool Reservoir Boundaries	Elevations from U.S. Army Corps of Engineers
Industrial Facilities Discharge (IFD) Sites	USEPA
Land Cover	1992-93 MoRAP Landcover Classification
Landfills	Missouri Department of Natural Resources, Air and Land Protection Division, Solid Waste Management Program
Mines - Coal	U.S. Bureau of Mines
Mines - Instream Gravel	Missouri Department of Conservation (MDC)
Mines - Lead	U.S. Bureau of Mines
Mines – All other	U.S. Bureau of Mines
Nonnative Species	Missouri Aquatic Gap Project - Predicted Species Distributions; Missouri Resource Assessment Partnership (MoRAP)
Permit Compliance System (PCS) Sites	USEPA; Ref: http://www/epa.gov/enviro
Resource Conservation and Recovery Information System (RCRIS) Sites	USEPA; Ref: http://www.epa.gov/enviro
Riparian Land Cover	MDC
Superfund National Priority List Sites	USEPA; Ref: http://www.epa.gov/enviro
TIGER Road Files	United States Department of Commerce, Bureau of the Census
Toxic Release Inventory (TRI) Sites	USEPA; Ref: http://www.epa.gov/enviro

Next, we generated statistics for 65 individual human stressors (e.g., percent urban, lead mine density, degree of fragmentation) for each of the 542 AES polygons in Missouri (Appendix 6.1). Forty eight of these metrics were generated with the EPA Analytical Tools Interface for Landscape Assessments (ATtILA, Version 3.0), which is an ArcView extension that allows users to easily calculate many common landscape metrics (USEPA 2004). Fourteen of the metrics were simply generated by sum or calculating point densities for data obtained from EPA BASINS 3.1 (USEPA 2001) or data explicitly developed for Missouri. Finally, three of the metrics, pertaining to hydrologic modification and network fragmentation, were categorical metrics and were visually determined by overlaying the high-pool reservoir boundaries onto the AES polygon boundaries. All of the metrics were calculated for each individual polygon and do not represent conditions within the overall watershed of each AES polygon.

Once the 65 metrics were calculated for every AES polygon, we used simple correlation analysis to reduce this overall set of metrics into a final set of 11, relatively uncorrelated ($r < 0.5$), measures of human disturbance (Table 6.2). Relativized rankings (range 1 to 4) were then developed for each of these 11 metrics (Table 6.2). These rankings are relative to the range of values obtained throughout the state of Missouri. If we were to use a different bounding area, some AES polygons would receive a different relative rank. A rank of 1 is indicative of relatively low disturbance for that particular metric, while a rank of 4 indicates a relatively high level of disturbance. These rankings were based on information contained within the literature or either quartiles or equal intervals when no empirical evidence on thresholds was available. For instance, rankings for percent urban were; 1: 0-5%, 2: 6-10%, 3: 11-20%, and 4: >20%, were based on the results of various studies that have examined the effects of urban land cover on the ecological integrity of stream ecosystems (Klein 1979; Osborne and Wiley 1988; Limburg et al. 1990; Booth 1991; Weaver and Garmen 1994; Booth and Jackson 1997; Wang et al. 2000). However, existing research for percent agriculture in the watershed has not identified clear thresholds, suggesting that there is a more or less continual decline in ecological integrity with each added percentage of agriculture in the watershed. For this measure of human stress we simply used four equal interval categories, 1: 0-25%, 2: 26-50%, 3: 51-75%, and 4: >75%.

Table 6.2. The 11 stressor metrics included in the Human Stressor Index (HSI) and the specific criteria used to define the four relative ranking categories for each metric that were used to calculate the HSI for each Aquatic Ecological System.

	Relative Ranks			
Metric	**1**	**2**	**3**	**4**
Number of Introduced Species	1	2	3	4-5
Percent Urban	0-5	5-10	11-20	>20
Percent Agriculture	0-25	26-50	51-75	>75
Density of Road-Stream Crossings (#/mi^2)	0-0.24	0.25-0.49	0.5-0.9	≥1
Population Change 1990-2000 (#/mi^2)	-42-0	0.1-14	15-45	>45
Degree of Hydrologic Modification and/or Fragmentation by Major Impoundments	1	2 or 3	4 or 5	6
Number of Federally Licensed Dams	0	1-9	10-20	>20
Density of Coal Mines (#/mi^2)	0	1-5	6-20	>20
Density of Lead Mines (#/mi^2)	0	1-5	6-20	>20
Density of Permitted Discharges (#/mi^2)	0	1-5	6-20	>20
Density of Confined Animal Feeding Operations (#/mi^2)	0	1-5	5-10	>10

Note: A major impoundment was defined as those that occur on streams classified as small or larger. The codes used to categorize the degree of hydrologic modification and/or fragmentation can be interpreted as follows.

1: No hydrologic alteration or fragmentation

2: Externally fragmented: obligate aquatic biota could reach one or more adjacent watersheds, but not the MO or MS Rivers without passing through a major impoundment

3: Hydrologically modified: included all innundated AES polygons and any area downstream of the dam known to have a significantly modified hydrologic regime

4: Both externally fragmented and hydrologically modified: includes those AES polygons that contain stream segments situated in the interceding area between two major impoundments on the same stream.

5: Isolated: obligate aquatic biota could not reach any adjacent watershed without passing through a major impoundment

6: Both Isolated and Hydrologically modified

The relativized rankings for each of the 11 metrics were then combined into a three number Human Stressor Index (HSI). The first number reflects the highest ranking across all 11 metrics (range 1 to 4). The last two numbers reflect the sum of the 11 metrics (range 11 to 44). This index allows you to evaluate both individual and cumulative impacts. For instance, a value of 418, indicates relatively low cumulative impacts (i.e., last two digits = 18 out of a possible 44), however, the first number is a 4, which indicates that one of the stressors is relatively high and potentially acting as a major human disturbance within the ecosystem.

6.5 Results

Figure 6.1 shows a map of the 542 AES polygons by the first value in the HSI (range 1-4). Over 95% of the AES polygons received a value of 3 or 4, indicating that the vast majority AESs are relatively threatened or impaired from at least one of the 11 human stressors included in the HSI. None of the AES polygons received the lowest value of 1 and just over twenty received a value of 2. Most of these AESs are located in the southcentral Ozarks within the Black/Current Ecological Drainage Unit (EDU), which is where most of the largest federal and state land holdings are within the Missouri. A closer examination of those AES polygons that received a value of 4 illustrates the diversity, especially the spatial diversity, of human stressors across Missouri (Figure 6.2). The greatest diversity occurs in the Ozark Aquatic Subregion, however, hydrologic modification/fragmentation due to both large reservoirs and smaller impoundments are the dominant stressors affecting the ecological integrity of riverine ecosystems in this Subregion. Row-crop agriculture is the dominant human stressor in both the Central Plains (CP) and Mississippi Alluvial Basin (MAB) Aquatic Subregions. Coal mine drainage is another potential stressor affecting the riverine ecosystems in the CP. Almost all of the AES polygons within the MAB received a value of 4 due to extremely high percentages of row-crop agriculture. Most of the AES polygons that contain multiple human stressors with a ranking of 4 occur within and adjacent to the major metropolitan areas in the state, such St. Louis, Kansas City, and Springfield.

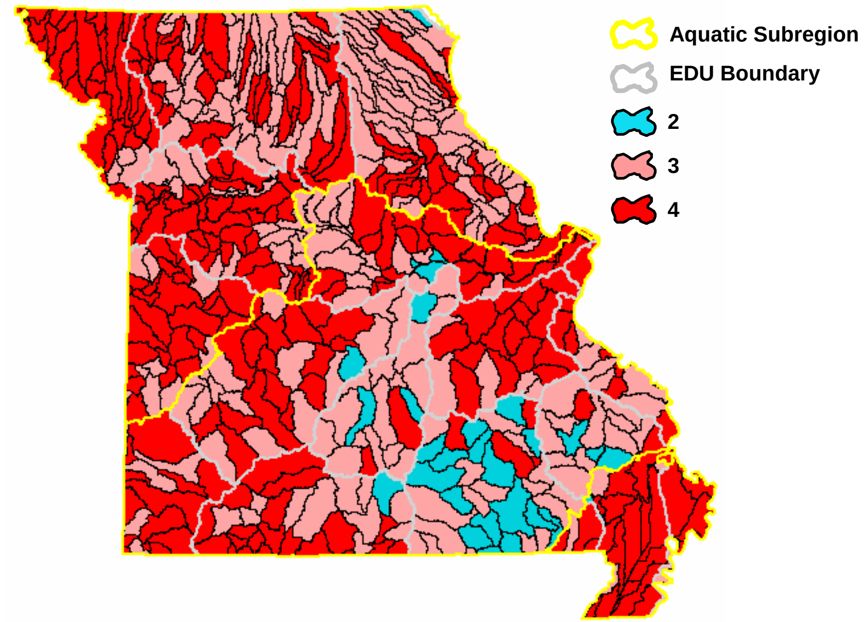


Figure 6.1. Map showing the first value in the Human Stressor Index (HSI) for each of the Aquatic Ecological Systems in Missouri. A value of 1 indicates a relatively low level of human disturbance, while a value of 4 indicates a relatively high level of disturbance. None of the AESs polygons received a value of 1.

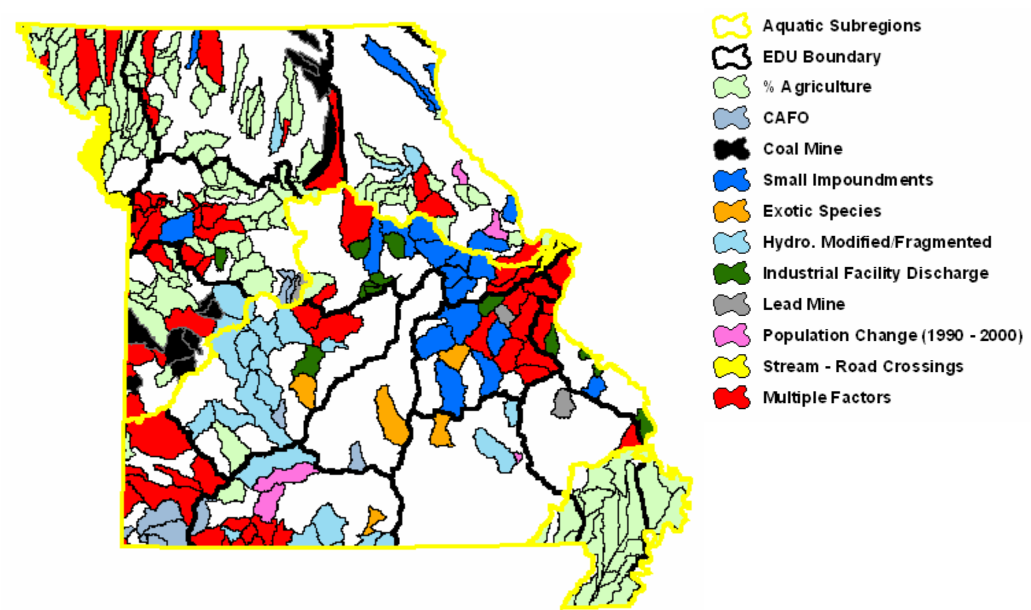


Figure 6.2. Map showing which AES polygons received a value of 4 for the first value in the Human Stressor Index, further broken down according to which specific human stressor was responsible for this high value.

When examining the spatial pattern of the last two values in the HSI, we find that cumulative disturbance tends to be highest in southwest Missouri and also in an east-west band throughout central Missouri (Figure 6.3). The AES polygons receiving the highest values for these last two digits of the HSI tend to fall within the most populated regions of the state. This same pattern holds when you examine the full 3-digit HSI across Missouri (Figure 6.4). Whether examining the individual components of the HSI or the overall index, the Black/Current EDU in the southcentral Ozarks stands out as the only major drainage in the state that is relatively undisturbed or ecologically intact. The fact that most of this EDU is within public ownership illustrates the importance of public lands to the long-term protection of freshwater biodiversity.

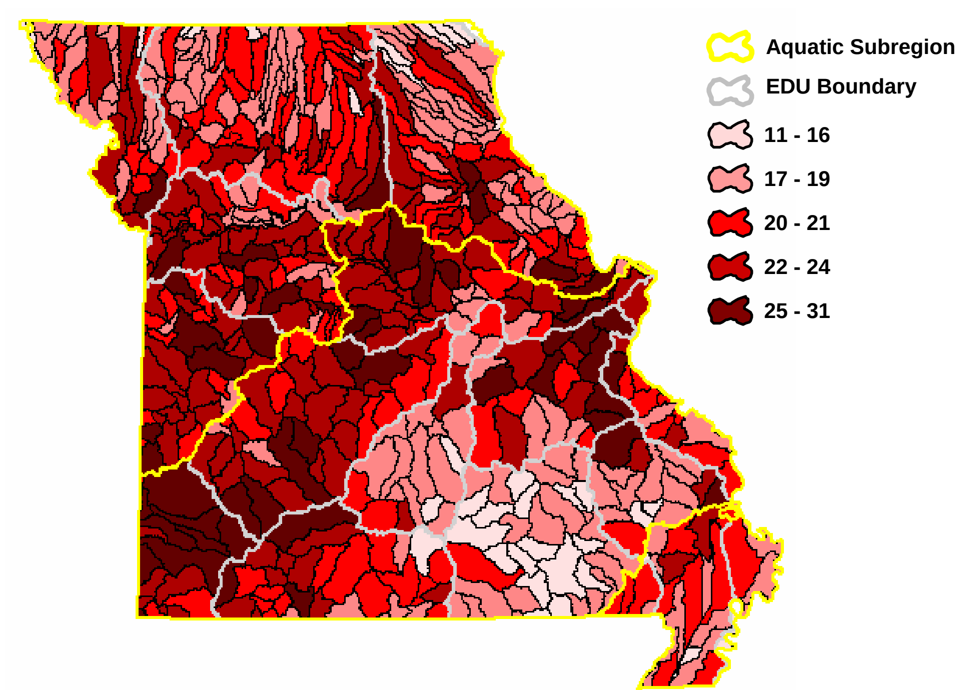


Figure 6.3. Map showing the last two values in the Human Stressor Index for each of the Aquatic Ecological Systems in Missouri. A value of 11 indicates an extremely low level of cumulative impact. The highest possible value in theory is a 44, however, because some of the 11 metrics used in the index are mutually exclusive (e.g., % urban and %agriculture), the highest obtainable value is unknown. The highest value in Missouri was 31. Basically, the higher the value for these last two digits, the higher degree of cumulative disturbance.

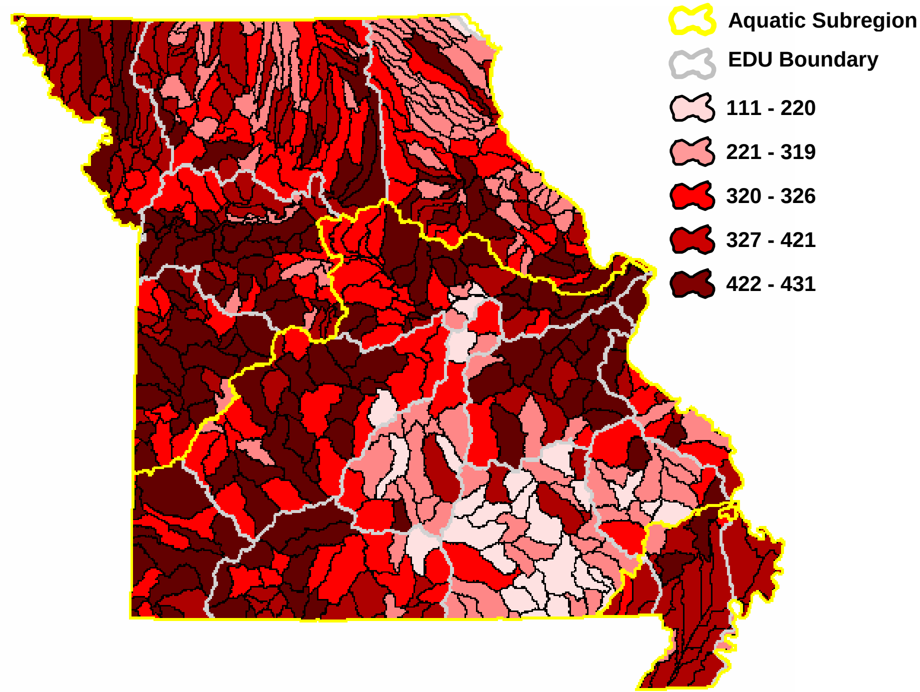


Figure 6.4. Map showing the composite Human Stressor Index (HSI) values for each Aquatic Ecological System in Missouri. The first number represents the highest value received across all 11 metrics included in the HSI, while the last two digits represent the sum of the scores received for each of the 11 metrics.

6.6 Discussion and Limitations

Accounting for human stressors within a GIS is an extremely difficult task. Given the complexity of the issue, nobody should expect perfect solutions (Rose 2000). Describing threats to the "health" of ecosystems with a handfull of metrics or indicators is similar to a general physical examination given to patient by a doctor. The metrics used in our Human Stressor Index (HSI) are of this general character. As we already stated, this index is an admittedly crude measure of human disturbance, however, it is well suited for a coarse-filter assessment since it does act as a “red flag.” Yet, the general metrics which make up this index are by no means a substitute for a more detailed assessment of ecosystem health.

There is no empirical evidence quantifying relations between our HSI, or the individual metrics, with the ecological integrity of the stream resources within Missouri. The HSI was developed as a relative measure of ecological integrity and to provide insight into the spatial distribution of the various human stressors across the Missouri landscape. However, there is a great deal of existing research, conducted throughout the United States and the world, documenting the influence of the various human stressors included in the HSI on the ecological integrity of freshwater ecosystems (Allan and Flecker 1993; Richter et al. 1997; USEPA 2000).

Considering the above, simply mapping the location and quantifying the abundance or density of particular set of stressors within a GIS is not enough. We must attribute GIS coverages, pertaining to human disturbance, with contextual information that enables users to more accurately account for the timing, magnitude, duration, and frequency of individual stressors and the combined cumulative effect on riverine ecosystems. There also needs to be substantially more research on how specific stressors influence the ecological integrity of receiving waters. Only through such quantification will we eventually be able to identify thresholds, like Wang et al. (2002) did for percent urban land use within a watershed, or develop models that account for the complex interaction among multiple stressors and their cumulative effects.

More than anything else, the data we generated for this portion of our project have the potential to educate the public and policy makers about the numerous threats facing the freshwater ecosystems in Missouri. Even we were surprised by the results, which showed that only a handful of watersheds remain relatively intact and the spatial distribution of the various stressors revealed a daunting task for stream resource managers in the state. We too often focus on the influence that urban and agricultural lands have on the integrity of freshwater ecosystems. However, as Figure 6.2 shows, there are a wide array of threats we must contend with in Missouri and that each part of the state has its own distinct combination of threats to go along with its own distinct freshwater ecosystems.

CHAPTER 7

The Aquatic GAP Analysis

Each state and region has naturally some characteristic territory and hence its own proper responsibility in the problem of preserving for the future the varied aquatic life of the continent.

Henry B. Ward, 1912
42nd Meeting of the American Fisheries Society

7.1 Background

As described in the general introduction of this report, the primary objective of GAP is to provide information on the distribution and status of several elements of biological diversity. For our project this is accomplished by first producing maps of: riverine ecosystem units at multiple scales (see Chapter 3), predicted distributions for selected animal species (see Chapter 4), and land stewardship and management status (see Chapter 5). Intersecting the land stewardship and management map with the distribution of the elements results in tables that summarize the area and percentage of total mapped distribution of each element in different land stewardship and management categories. The data are provided in a format that allows users to query the representation of each element in different management status categories, as appropriate to their own management objectives. This forms the basis of GAP's mission to provide landowners and managers with the information necessary to conduct informed policy development, planning, and management for biodiversity maintenance.

Although GAP "seeks to identify habitat types and species not adequately represented in the current network of biodiversity management areas" (GAP Handbook, Preface, Version 1, pg. I), it is unrealistic to create a standard definition of "adequate representation" for either habitat or species (Noss et al. 1995). A practical solution to this problem is to report both percentages and absolute area of each biodiversity element within each management status category and allow the user to determine which elements are adequately represented based on detailed studies of the ecology, population viability assessments as well as studies of the spatial and temporal dimensions of ecological processes.

Clearly, opinions will differ among users, but this disagreement is an issue of policy, not scientific analysis. We have, however, provided a breakdown along six levels of representation (0, 0.1-<1%, 1-<10%, 10-<20%, 20-<50%, and $\geq$50%). The zero and <1% levels indicate those elements with none or essentially none of their distribution in a protected status, while levels 10%, 20% and 50% have been recommended in the literature as necessary amounts of conservation (Odum and Odum 1972, Specht et al. 1974, Ride 1975, Miller 1984, Noss 1991, Noss and Cooperrider 1994). The network of Conservation Data Centers (CDCs) and Natural Heritage Programs (NHPs), established cooperatively by The Nature Conservancy and various state agencies, maintains detailed databases on the locations of rare elements of biodiversity. GAP cooperatively uses these data to develop predicted distributions of potentially suitable habitat for these elements, which may be valuable for identifying research needs and preliminary considerations for restoration or reintroduction. Conservation of such elements, however, is best accomplished through more detailed "fine-filter" assessments. It is not the role of GAP to duplicate or disseminate Heritage Program or CDC Element Occurrence Records. Users interested in more specific information about the location, status, and ecology of populations of such species are directed to their state Heritage Program or CDC.

7.2 Basic Elements of Our Gap Analysis

Generally, the GAP management status categories of 1 and 2 are considered to have reasonably secure conservation provisions that benefit biodiversity and our gap analyses focus on these categories. Our gap analyses also focus on local management status of individual stream segments versus the percent of the watershed or upstream network within status 1 or 2 lands. The reason for this decision is we believe that, in terms of reserve design, local public ownership is a necessary first step for the long-term maintenance of riverine biodiversity since even the most ambitious of watershed management efforts (e.g., 90% of watershed in status 1 or 2 lands) can be undone by local human disturbances to the stream segments that are the focus of conservation. Another reason for our decision to focus on local management status pertains to the lack of empirical data addressing the question of, "How much is enough?" Is 25, 50, or 75% of a watershed in status 1 or 2 lands sufficient to ensure long-term protection? Such thresholds must be identified before they can be included in any gap analysis.

Our gap analyses quantify representation of both abiotic and biotic elements of biodiversity. For the abiotic elements we generated statistics to address two fundamental questions;

- How well are the various stream types (Valley Segment Types) represented within the existing matrix of public lands set aside for long term maintenance of biodiversity (Status 1 or 2 lands)?

- How well are the various watershed types (Aquatic Ecological System Types) represented within the existing matrix of public lands set aside for long term maintenance of biodiversity?

By addressing these questions we are attempting to assess the representation of the various riverine habitats across the Missouri landscape, which may prove more useful than assessing representation of individual species (Angermeier and Schlosser 1995). Our analyses for the biotic elements (fish, mussel and crayfish species) follow those used in previous GAP projects dealing with terrestrial vertebrates. However, our statistics are presented in terms of length, not area, since we are dealing with linear and not polygonal data. Furthermore, we also examine the number of distinct locations in which each species is represented in status 1 or 2 lands, which is further explained in the species analysis section below.

The conservation status statistics for these biodiversity elements are examined from a statewide perspective and then further examined within the context of our Aquatic Subregions and Ecological Drainage Units. This was done in order to examine the broadscale representation of biodiversity throughout these ecological units. These broader-scale assessments provide an important and more holistic context for biodiversity conservation than the evaluation of individual elements of biodiversity across the state.

7.3 Analysis of Abiotic Elements

Valley Segment Type Analysis

To assess the conservation status of the various stream types within Missouri, we used a five-variable Valley Segment Type (VST) code that included five fundamental parameters; temperature (2 classes), stream size (4 classes), flow (2 classes), geology (5 classes), and relative gradient (3 classes). Within our 1:100,000 Valley Segment Coverage there are 196 distinct VSTs based on these five parameters. However, many of these VSTs are spurious types due to inherent errors in the data and errors that occurred due to the fact that the attributes were generated with various source data that were developed at different spatial scales (1:24,000 to 1:500,000). Of course, some are also rare stream types, yet we believe that our coarse-filter assessment should initially focus on the most characteristic/common stream types within the state.

To reduce this overall set of 196 VSTs we performed a set of queries on the attribute table of the VST coverage. First, we selected all coldwater VSTs and those flowing through igneous geology that had greater than 10 km of total length within the state. Next, from the remaining set of VSTs, we selected those that had greater than 100 km of total length within the state. Coldwater streams and those flowing through igneous geology are rare stream types in Missouri (Pflieger 1989), which is why we used a lower inclusion threshold for these stream types. Our queries removed 122 of the VSTs, which represented just 1.5% of the total length of stream in the state. Consequently, our analyses focus on 74 distinct VSTs that represent 98.5% of the total length of stream in Missouri. As expected, the number of VSTs decreases rapidly with stream size, with 29 headwater, 23 creek, 14 small river, and 8 large river VSTs.

<u>Results</u>

Lands classified as GAP management status 1 or 2 contain less than 1% (0.8%) of the total length of stream within Missouri (Figure 7.1). Most of these lands, and therefore stream miles flowing within status 1 or 2, are situated in the Ozark Aquatic Subregion (1,185 km) (Table 7.1). However, this still only represents just over 1% of the total length of stream that occurs within this Subregion.

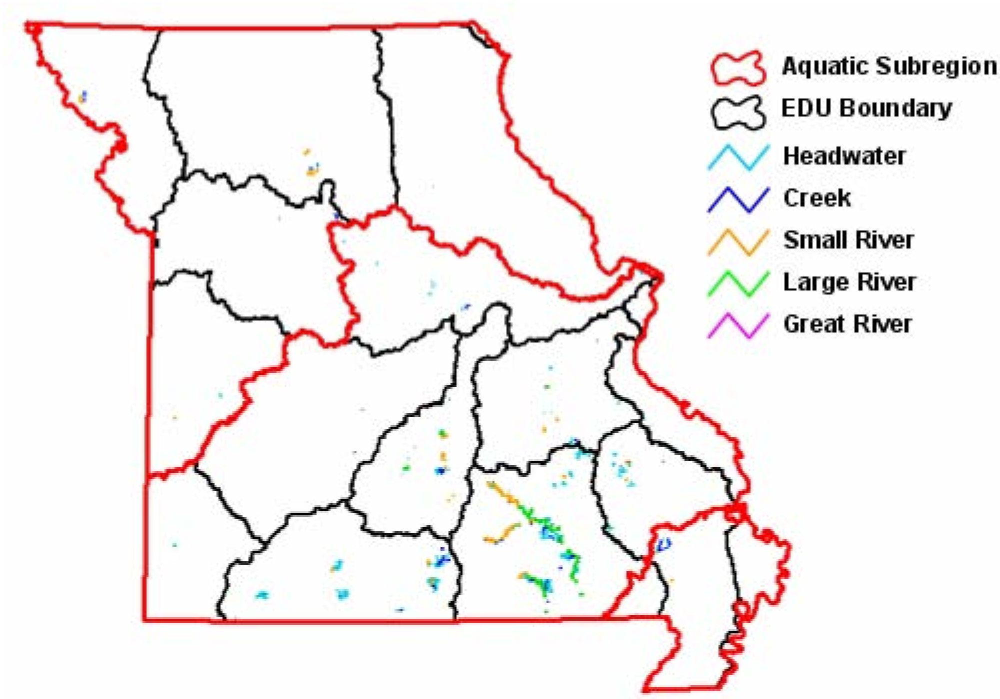


Figure 7.1. Map showing the distribution of stream segments with more than 50% of their length within status 1 or 2 lands, broken down by stream size.

Table 7.1. Total miles and percent of total miles of stream that are flowing within status 1 or 2 lands for each Aquatic Subregion.

		Status 1 or 2	
Subregion	**Total Km**	**Kilometers**	**Percent**
Central Plains	70,243	97	0.14
Ozarks	93,230	1,185	1.27
Mississippi Alluvial Basin	8,929	60	0.67

From a statewide perspective, a relatively high percentage (77%) of the 74 VSTs are presently represented in status 1 or 2 lands (Table 7.2). This statewide percentage is nearly matched in the Ozark Aquatic Subregion, however, a considerably lower percentage of the VSTs that occur within the CP and the MAB have been captured in status 1 or 2 lands (Table 7.2). As would be expected from Figure 7.1, when we examined the representation of VSTs by the various Ecological Drainage Units (EDUs) we find a great deal of variation, especially within the Ozarks and the MAB (Table 7.3, Figure 7.2). On average, EDUs within the CP had 11.4% of the VSTs represented in status 1 or 2 lands, compared with an average of 28.3% in the Ozarks and 20.9% in the MAB. Within the Ozarks there was a high degree of variation, ranging from a single VST represented in the Neosho EDU (3.6%) to 34 VSTs represented in the Black/Current EDU (63.0%). There was also a high degree of variation among the three EDUs within the MAB. Forty percent of the VSTs within the St. Francis/Little EDU were represented in status 1 or 2 lands, compared with 14.7% in Black/Cache and only 8.3 in the St. John's Bayou. Like the Neosho, only a single VST was represented in these last two EDUs.

Table 7.2. Statistics showing the total number of VSTs in Missouri and each Aquatic Subregion with the corresponding number and percent of these totals that are captured in status 1 or 2 lands.

		Status 1 or 2	
State/Subregion	Total VSTs	Number	Percent
Statewide	74	57	77
Central Plains	45	14	31
Ozarks	65	49	75
Mississippi Alluvial Basin	30	13	43

Table 7.3. Statistics showing the total number of VSTs within each Ecological Drainage Unit (EDU) and the corresponding number and percent of these totals that are captured in status 1 or 2 lands. Average percentages are also provided for each Aquatic Subregion.

Subregion	EDU	Total VSTs	Number in Status 1 or 2	Percent in Status 1 or 2	Average Percent
Central Plains	Blackwater/Lamine	42	3	7.1	
	Grand/Chariton	41	5	12.2	
	Nishnabotna/Platte	26	3	11.5	
	Osage/South Grand	33	3	9.1	
	Cuivre/Salt	35	6	17.1	**11.4**
Ozarks	Black/Current	54	34	63.0	
	Neosho	28	1	3.6	
	Gasconade	42	19	45.2	
	Apple/Joachim	37	5	13.5	
	Meramec	45	9	20.0	
	Moreau/Loutre	38	9	23.7	
	Osage	40	4	10.0	
	Upper St. Francis/Castor	49	13	26.5	
	White	43	21	48.8	**28.3**
Mississippi Alluvial Basin	St. Francis/Little	30	12	40.0	
	St. Johns Bayou	12	1	8.3	
	Black/Cache	7	1	14.3	**20.9**

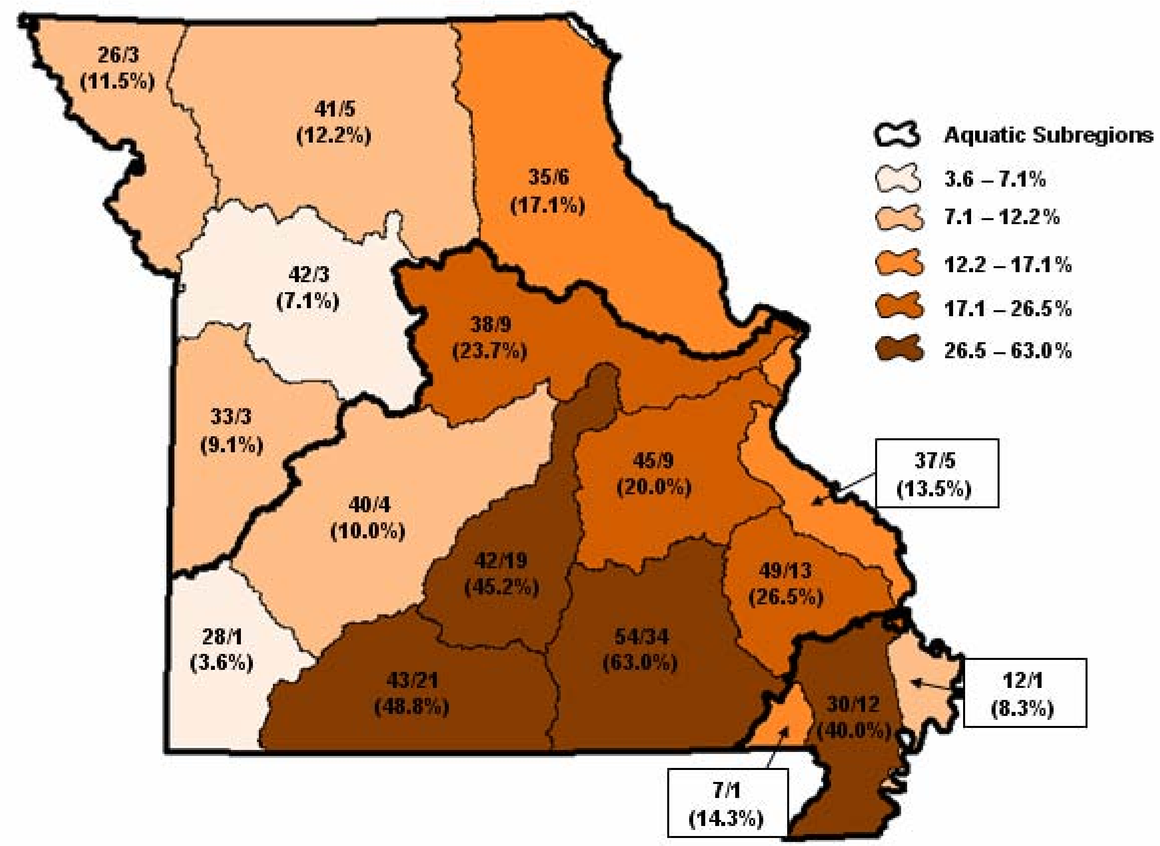


Figure 7.2. Map showing the total number of Valley Segment Types (VSTs) within each Ecological Drainage Unit (EDU) with the corresponding number and percent of those VSTs that are represented in status 1 or 2 lands.

Table 7.4 provides the conservation status statistics for each the 74 distinct VSTs examined in our analyses. While useful for detailed evaluations of the representation of the various stream types in Missouri, a more general but informative evaluation can be made by examining representation by each of the five parameters that comprise the VSTs. The results of these analyses are provided in Table 7.5. The most striking results are that coldwater streams and streams flowing through igneous geology have a significantly higher percentage of their total length represented in status 1 or 2 lands. These stream types are certainly distinctive and rare, yet are also some of the least biologically diverse stream types in the state (Pflieger 1989). Furthermore, most of the coldwater streams contain highly managed populations of nonnative rainbow trout and brown trout.

Table 7.4. Conservation status statistics for 74 distinct Valley Segment Types (VST) in Missouri. Table shows the individual elements of each VST, the total length of each VST in the state, and both the total and percent of total length flowing within Status 1 or 2 lands. The table is sorted by temperature and then stream size, with distinct cell boundaries between these attribute classes to aid in the interpretation of the data.

VST Code	Temperature	Stream Size	Flow	Geology	Relative Gradient	Total Length (Km)	Length in Status 1 or 2	Percent in Status 1 or 2
11121	Cold Water	Headwater	Permanent Flow	Limestone/Dolomite	Low Gradient	12.97	0	0.00
11223	Cold Water	Headwater	Intermittent Flow	Limestone/Dolomite	High Gradient	22.48	0	0.00
12121	Cold Water	Creek	Permanent Flow	Limestone/Dolomite	Low Gradient	124.98	0	0.00
12122	Cold Water	Creek	Permanent Flow	Limestone/Dolomite	Moderate Gradient	88.70	0	0.00
12123	Cold Water	Creek	Permanent Flow	Limestone/Dolomite	High Gradient	50.96	1.89	3.71
13121	Cold Water	Small River	Permanent Flow	Limestone/Dolomite	Low Gradient	12.07	1.43	11.84
13122	Cold Water	Small River	Permanent Flow	Limestone/Dolomite	Moderate Gradient	55.32	12.73	23.02
13123	Cold Water	Small River	Permanent Flow	Limestone/Dolomite	High Gradient	84.57	13.45	15.90
13142	Cold Water	Small River	Permanent Flow	Sandstone	Moderate Gradient	12.90	0	0.00
14122	Cold Water	Large River	Permanent Flow	Limestone/Dolomite	Moderate Gradient	36.90	0	0.00
14123	Cold Water	Large River	Permanent Flow	Limestone/Dolomite	High Gradient	35.77	22.59	63.15
14143	Cold Water	Large River	Permanent Flow	Sandstone	High Gradient	19.87	7.24	36.46
21010	Warm Water	Headwater	Unknown Flow	Alluvium	NA(MAB Subregion)	3529.95	0.12	0.00
21011	Warm Water	Headwater	Unknown Flow	Alluvium	Low Gradient	261.57	0	0.00
21110	Warm Water	Headwater	Permanent Flow	Alluvium	NA(MAB Subregion)	219.46	2.90	1.32
21111	Warm Water	Headwater	Permanent Flow	Alluvium	Low Gradient	413.70	9.62	2.32
21121	Warm Water	Headwater	Permanent Flow	Limestone/Dolomite	Low Gradient	6995.46	53.94	0.77
21122	Warm Water	Headwater	Permanent Flow	Limestone/Dolomite	Moderate Gradient	1445.96	2.71	0.19
21123	Warm Water	Headwater	Permanent Flow	Limestone/Dolomite	High Gradient	546.46	6.24	1.14
21131	Warm Water	Headwater	Permanent Flow	Igneous	Low Gradient	17.98	1.32	7.36
21132	Warm Water	Headwater	Permanent Flow	Igneous	Moderate Gradient	17.26	4.34	25.13
21133	Warm Water	Headwater	Permanent Flow	Igneous	High Gradient	31.89	1.25	3.90
21141	Warm Water	Headwater	Permanent Flow	Sandstone	Low Gradient	1066.36	14.99	1.41
21142	Warm Water	Headwater	Permanent Flow	Sandstone	Moderate Gradient	252.40	0.25	0.10
21210	Warm Water	Headwater	Intermittent Flow	Alluvium	NA(MAB Subregion)	1240.93	1.38	0.11
21211	Warm Water	Headwater	Intermittent Flow	Alluvium	Low Gradient	1742.88	17.08	0.98
21212	Warm Water	Headwater	Intermittent Flow	Alluvium	Moderate Gradient	151.18	2.13	1.41
21220	Warm Water	Headwater	Intermittent Flow	Limestone/Dolomite	NA(MAB Subregion)	146.87	0.00	0.00
21221	Warm Water	Headwater	Intermittent Flow	Limestone/Dolomite	Low Gradient	36863.30	68.10	0.18

Table 7.4. Continued.

VST Code	Temperature	Stream Size	Flow	Geology	Relative Gradient	Total Length (Km)	Length in Status 1 or 2	Percent in Status 1 or 2
21222	Warm Water	Headwater	Intermittent Flow	Limestone/Dolomite	Moderate Gradient	32227.41	65.93	0.20
21223	Warm Water	Headwater	Intermittent Flow	Limestone/Dolomite	High Gradient	22790.69	266.86	1.17
21231	Warm Water	Headwater	Intermittent Flow	Igneous	Low Gradient	32.61	0.00	0.00
21232	Warm Water	Headwater	Intermittent Flow	Igneous	Moderate Gradient	91.50	2.02	2.20
21233	Warm Water	Headwater	Intermittent Flow	Igneous	High Gradient	344.08	36.36	10.57
21241	Warm Water	Headwater	Intermittent Flow	Sandstone	Low Gradient	6366.05	54.04	0.85
21242	Warm Water	Headwater	Intermittent Flow	Sandstone	Moderate Gradient	7201.91	61.77	0.86
21243	Warm Water	Headwater	Intermittent Flow	Sandstone	High Gradient	4462.02	98.33	2.20
21251	Warm Water	Headwater	Intermittent Flow	Clay	Low Gradient	143.30	0.00	0.00
21252	Warm Water	Headwater	Intermittent Flow	Clay	Moderate Gradient	160.80	0.00	0.00
22010	Warm Water	Creek	Unknown Flow	Alluvium	NA(MAB Subregion)	1272.11	2.21	0.17
22011	Warm Water	Creek	Unknown Flow	Alluvium	Low Gradient	112.09	4.70	4.19
22110	Warm Water	Creek	Permanent Flow	Alluvium	NA(MAB Subregion)	153.59	16.98	11.05
22111	Warm Water	Creek	Permanent Flow	Alluvium	Low Gradient	545.93	9.07	1.66
22121	Warm Water	Creek	Permanent Flow	Limestone/Dolomite	Low Gradient	8764.44	11.45	0.13
22122	Warm Water	Creek	Permanent Flow	Limestone/Dolomite	Moderate Gradient	4435.35	15.21	0.34
22123	Warm Water	Creek	Permanent Flow	Limestone/Dolomite	High Gradient	3237.06	42.09	1.30
22132	Warm Water	Creek	Permanent Flow	Igneous	Moderate Gradient	30.26	0.00	0.00
22133	Warm Water	Creek	Permanent Flow	Igneous	High Gradient	50.79	7.67	15.11
22141	Warm Water	Creek	Permanent Flow	Sandstone	Low Gradient	1060.32	0.30	0.03
22142	Warm Water	Creek	Permanent Flow	Sandstone	Moderate Gradient	648.15	20.54	3.17
22143	Warm Water	Creek	Permanent Flow	Sandstone	High Gradient	498.07	4.39	0.88
22210	Warm Water	Creek	Intermittent Flow	Alluvium	NA(MAB Subregion)	138.25	0.00	0.00
22211	Warm Water	Creek	Intermittent Flow	Alluvium	Low Gradient	163.14	0.00	0.00
22221	Warm Water	Creek	Intermittent Flow	Limestone/Dolomite	Low Gradient	1622.26	1.56	0.10
22222	Warm Water	Creek	Intermittent Flow	Limestone/Dolomite	Moderate Gradient	1552.01	7.64	0.49
22223	Warm Water	Creek	Intermittent Flow	Limestone/Dolomite	High Gradient	1748.04	15.88	0.91
22241	Warm Water	Creek	Intermittent Flow	Sandstone	Low Gradient	252.76	1.14	0.45
22242	Warm Water	Creek	Intermittent Flow	Sandstone	Moderate Gradient	326.00	0.00	0.00
22243	Warm Water	Creek	Intermittent Flow	Sandstone	High Gradient	298.58	18.25	6.11
23010	Warm Water	Small River	Unknown Flow	Alluvium	NA(MAB Subregion)	418.50	1.79	0.43
23111	Warm Water	Small River	Permanent Flow	Alluvium	Low Gradient	408.03	10.71	2.62
23121	Warm Water	Small River	Permanent Flow	Limestone/Dolomite	Low Gradient	3718.75	29.39	0.79

Table 7.4. Continued.

VST Code	Temperature	Stream Size	Flow	Geology	Relative Gradient	Total Length (Km)	Length in Status 1 or 2	Percent in Status 1 or 2
23122	Warm Water	Small River	Permanent Flow	Limestone/Dolomite	Moderate Gradient	3593.68	96.31	2.68
23123	Warm Water	Small River	Permanent Flow	Limestone/Dolomite	High Gradient	2108.15	23.40	1.11
23132	Warm Water	Small River	Permanent Flow	Igneous	Moderate Gradient	27.72	0.00	0.00
23133	Warm Water	Small River	Permanent Flow	Igneous	High Gradient	20.23	5.18	25.63
23141	Warm Water	Small River	Permanent Flow	Sandstone	Low Gradient	418.33	3.04	0.73
23142	Warm Water	Small River	Permanent Flow	Sandstone	Moderate Gradient	409.16	11.83	2.89
23143	Warm Water	Small River	Permanent Flow	Sandstone	High Gradient	199.04	1.01	0.51
24110	Warm Water	Large River	Permanent Flow	Alluvium	NA(MAB Subregion)	260.76	0.00	0.00
24121	Warm Water	Large River	Permanent Flow	Limestone/Dolomite	Low Gradient	1140.67	6.63	0.58
24122	Warm Water	Large River	Permanent Flow	Limestone/Dolomite	Moderate Gradient	753.76	38.27	5.08
24123	Warm Water	Large River	Permanent Flow	Limestone/Dolomite	High Gradient	860.47	67.64	7.86
24142	Warm Water	Large River	Permanent Flow	Sandstone	Moderate Gradient	130.99	2.91	2.22

Table 7.5. Conservation status statistics for the individual parameters used to classify distinct Valley Segment Types. Table shows the total length of stream that occurs in the state for each parameter, and both the total and percent of total length flowing within Status 1 or 2 lands. *NOTE: these statistics only pertain to the common/characteristic VSTs, not all streams.*

Temperature	Total Length (km)	Km in Status 1 or 2	Percent in Status 1 or 2
Cold	557.5	59.3	10.6
Warm	171,700.1	1248.8	0.7
Stream Size	**Total Length (km)**	**Length in Status 1 or 2**	**Percent in Status 1 or 2**
Headwater	128,799.4	771.6	0.6
Creek	27,173.8	180.9	0.7
Small River	11,486.5	210.3	1.8
Large River	3,239.19	145.3	4.49
Flow	**Total Length (km)**	**Length in Status 1 or 2**	**Percent in Status 1 or 2**
Intermittent	120,089.1	718.5	0.6
Perennial	46,574.33	580.9	1.25
Geology	**Total Length (km)**	**Length in Status 1 or 2**	**Percent in Status 1 or 2**
Alluvium	12,404.3	78.7	0.6
Clay	304.1	0	0.0
Igneous	664.3	58.1	8.8
Limestone/Dolomite	135,075.5	871.3	0.7
Sandstone	23,622.9	300.0	1.3
Relative Gradient	**Total Length (km)**	**Length in Status 1 or 2**	**Percent in Status 1 or 2**
Low	72,260.0	298.5	0.4
Moderate	53,649.3	344.6	0.6
High	37,409.2	639.7	1.7

Another interesting result is the positive association between stream size and percent representation in status 1 or 2 lands. Large rivers have the highest percent of their total length represented in these lands, however, a closer examination of the more detailed data provided in Table 7.4, shows that most of the 145.3 km of large river that are flowing through status 1 or 2 lands is classified as coldwater (29.8 km). Consequently, 20% of the total length of the large rivers that are flowing within status 1 or 2 lands are coldwater streams. Yet, as Table 7.5 shows there are significantly more total kilometers of warmwater streams represented in status 1 or 2 lands. Table 7.5 also shows that a significantly higher percentage of the relatively high gradient streams are captured compared to lower gradient streams. This illustrates that fact that much of the public lands in Missouri are situated in the relatively rugged and higher elevation landscapes that are less productive and more difficult to develop.

Figure 7.3 shows that approximately 25% of the VSTs that occur in each stream size class are not currently represented in status 1 or 2 lands. This figure also shows that most of the VSTs have between 1 and 10 km captured in these lands. For instance, three of the eight large river VSTs have between 1 and 10 km of their length in status 1 or 2 lands. There are no creek VSTs that have greater than 50 km represented.

However, seven of the 29 headwater VSTs (24%) have greater than 50 km flowing within status 1 or 2 lands.

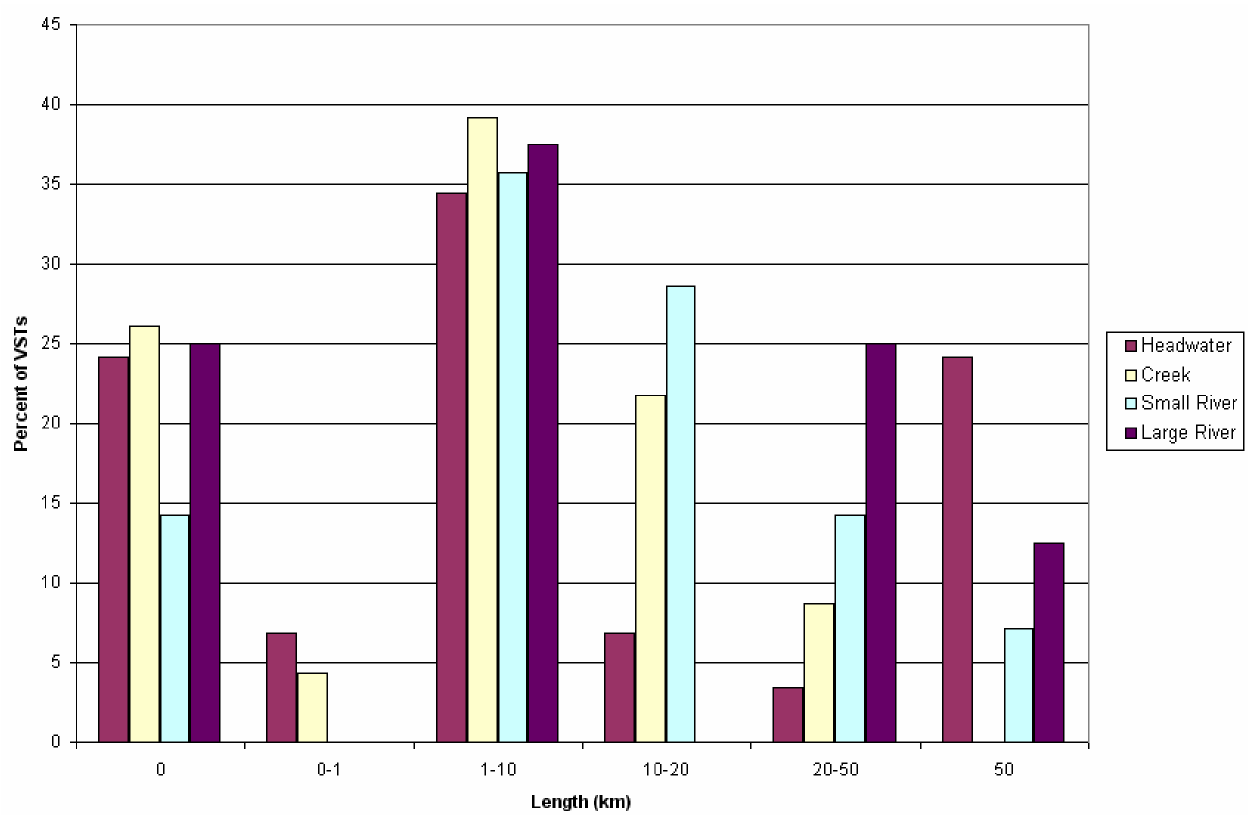


Figure 7.3. Bar chart showing the percentage of Valley Segment Types (VSTs), for each stream size class, that occur in six levels of representation (status 1 or 2 lands) by length. For example, of the eight distinct large river VSTs; 2 (25%) are not represented in any status 1 or 2 lands, 0 have between zero and 1 km, 3 (37.5%) have between 1 and 10 km, 0 (0.0%) have between 10 and 20 km, 2 (25%) have between 20 and 50 km, and 1 (12.5%) has greater than 50 km represented.

Aquatic Ecological System Type Analysis

Each individual Aquatic Ecological System (AES) contains three stream size classes; a) headwater, creek, and small river, b) headwater, creek and large river, or c) headwater, creek, and great river. As Pflieger (1989) and our data show, the biological assemblages that occur within these various stream sizes are dramatically different. In addition, many species collectively utilize the distinct habitats of these different stream size classes in order to successfully meet various life-history requirements (Schlosser and Angermeier 1995). Consequently, to assess the representation of these broader-scale ecological units we determined that, at an absolute minimum, each of the three size classes within a given AES polygon should be represented in status 1 or 2 lands before it could be considered effectively represented. Therefore, our analyses initially focus on identifying those individual AESs that have all three stream size classes represented in status 1 or 2 lands. We then apply more "stringent" and, we believe, more ecologically meaningful criteria. Specifically, we then identified those individual AES polygons have all three stream size classes, that occur within their boundary, represented as an interconnected matrix in status 1 or 2 lands. Finally, using our Human Stressor Index and other available geospatial data on human stressors, we examined the ecological integrity of the AESs, meeting the above criteria, in an effort to assess whether or not these units are effectively providing long-term maintenance of freshwater biodiversity.

Results

Only 19(3.5%) of the 542 individual AESs within Missouri have all three stream size classes represented in status 1 or 2 lands (Table 7.6; Figure 7.4). These 19 individual AESs represent just 6(15%) of the 39 distinct AES-Types that occur in the state. Three of the 13 (23%) AES-Types that occur in the Central Plains are represented using these criteria, compared with 2 of 25 (8%) in the Ozarks, and 1 of 11 (9%) in the Mississippi Alluvial Basin. In most instances, only a single AES of a given type is represented. However, the Jacks Fork AES-Type has 13 (32.5%) individual AESs with all stream size classes represented in status 1 or 2 lands. While representing multiple examples of a given ecosystem type is often desirable for long-term maintenance of biodiversity, this instance illustrates an extreme case of redundancy in representation. East Locust Creek, which is the most common watershed type in the Central Plains, is the only other AES-Type to be represented more than once, with 2 of the 59 individual AESs of this type having all three stream sizes captured in status 1 or 2 lands.

Table 7.6. Conservation status statistics for each Aquatic Ecological System (AES) Type within each Aquatic Subregion. Table shows the total number of individual AESs of each Type that occur in each Subregion along with the number and percent that have all three stream size classes represented in status 1 or 2 lands.

Central Plains			
AES Type	**Total Individual AESs**	**Number Represented In Status 1 or 2**	**% Represented in Status 1 or 2**
Boeuf Creek	8	0	0.00
Clear Creek	18	0	0.00
East Locust Creek	59	**2**	**3.39**
Honey Creek	19	0	0.00
Lick Creek	40	0	0.00
Middle River	4	0	0.00
Moniteau Creek	3	0	0.00
Ramsey Creek	10	**1**	**10.00**
Rock Creek	23	**1**	**4.35**
Sampson Creek	49	0	0.00
South Deepwater Creek	43	0	0.00
Tavern Creek	1	0	0.00
Upper Cuivre River	4	0	0.00

Table 7.6. Continued.

Table 7.6. Continued.

Ozarks			
AES Type	**Total Individual AESs**	**Total Represented In Status 1 or 2**	**% Represented in Status 1 or 2**
Beaver Creek	7	0	0.00
Big Creek	2	0	0.00
Boeuf Creek	30	0	0.00
Bull Creek	12	0	0.00
Clear Creek	5	0	0.00
Crowley's Ridge	1	0	0.00
Dry Fork	8	0	0.00
Finley Creek	15	0	0.00
Indian Creek	9	0	0.00
Jacks Fork	40	**13**	**32.50**
Lick Creek	1	0	0.00
Little St. Francis River	8	**1**	**12.50**
Lower Meramec	1	0	0.00
Middle River	10	0	0.00
Upper Big River	6	0	0.00
Upper Little Sac	8	0	0.00
Moniteau Creek	13	0	0.00
Ramsey Creek	7	0	0.00
Rock Creek	1	0	0.00
South Deepwater Creek	3	0	0.00
Spring Creek	4	0	0.00
Spring River	4	0	0.00
Tavern Creek	19	0	0.00
Upper Big Piney	9	0	0.00
Upper Spring River/Neosho	3	0	0.00
Mississippi Alluvial Basin			
AES Type	**Total Individual AESs**	**Total Represented In Status 1 or 2**	**% Represented in Status 1 or 2**
Cane Creek	4	0	0.00
Chaffee	2	0	0.00
Charleston	2	0	0.00
Gideon	1	0	0.00
Hayti	4	0	0.00
Senath	2	0	0.00
Crowley's Ridge	7	**1**	**14.29**
Little River	1	0	0.00
St. Johns Diversion Ditch	1	0	0.00
West Ditch	2	0	0.00
Wilkerson Ditch	1	0	0.00

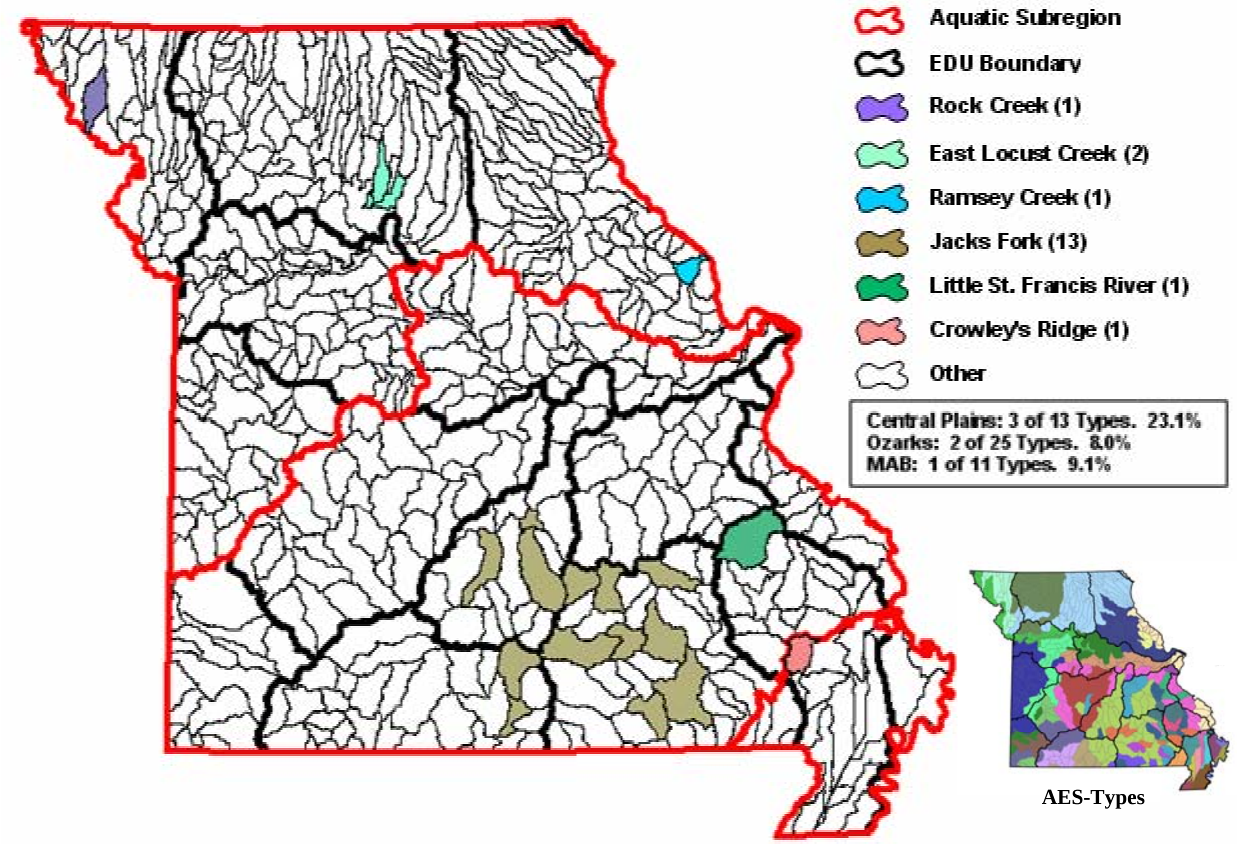


Figure 7.4. Map showing the individual Aquatic Ecological Systems (AESs) that have all stream size classes within their boundary represented in status 1 or 2 lands. The numbers, in parentheses, next to the name of the six AES-Types are the number of individual AESs of that type which are represented using the above criteria. Inset box shows the number and percentage of the AES-Types that represented within each Aquatic Subregion. Inset map shows all AES-Types for context.

Figure 7.4 clearly illustrates that none of the EDUs have more than a single AES-Type represented within their boundary, despite the fact that some have multiple individual AESs represented. For instance, within the Black/Current EDU there are eight individual AESs that have all size classes represented in status 1 or 2 lands. However, all eight of these AESs are classified as the Jacks Fork AES-Type (Figure 7.4 and Table 7.7). Therefore, eight of the nine AES-Types that occur within this EDU are not represented using these criteria. This redundancy in representation also occurs in three other EDUs; Grand/Chariton, Gasconade, and White.

Table 7.7. Conservation status statistics for each Aquatic Ecological System (AES) Type within each Ecological Drainage Unit (EDU). Table shows the total number of AES Types that occur in each EDU along with the number and percent that have all three stream size classes represented in status 1 or 2 lands.

Subregion	EDU	Total AES-Types	Number Represented	Percent Represented
Central Plains	Blackwater/Lamine	9	0	0.0
	Cuivre/Salt	4	1	25.0
	Grand/Chariton	3	1	33.3
	Nishnabotna/Platte	4	1	25.0
	Osage/South Grand	2	0	0.0
Ozarks	Apple/Joachim	3	0	0.0
	Black/Current	9	1	11.1
	Gasconade	6	1	16.7
	Meramec	7	0	0.0
	Moreau/Loutre	7	0	0.0
	Neosho	5	0	0.0
	Osage	8	0	0.0
	Uppper St.Francis/Castor	7	1	14.3
	White	5	1	20.0
Mississippi Alluvial Basin	Black/Cache	1	0	0.0
	St. Francis/Little	7	1	14.3
	St. Johns Bayou	3	0	0.0

When we apply slightly more stringent criteria to assess the representation of AESs we find seven of the 19, depicted in Figure 7.4, do not have all of the size classes represented in status 1 or 2 lands as an interconnected complex (Figure 7.5). This illustrates the fragmented nature of public land holdings in Missouri. When we apply even more stringent criteria and assess the general ecological integrity of those 12 AESs depicted in Figure 7.5, we find that only 4 (33%) can be considered relatively undisturbed or ecologically intact (Figure 7.6). All four of these occur within the Black/Current EDU and furthermore all four are represent the same AES-Type (Jacks Fork). This broader-scale assessment of ecosystem representation paints a bleak picture for freshwater biodiversity conservation in Missouri. Our failure to examine and conserve our freshwater resources within these broader contexts has likely contributed to the decline of freshwater biodiversity in Missouri, which has many freshwater species exhibiting significant declines (Horner 2005).

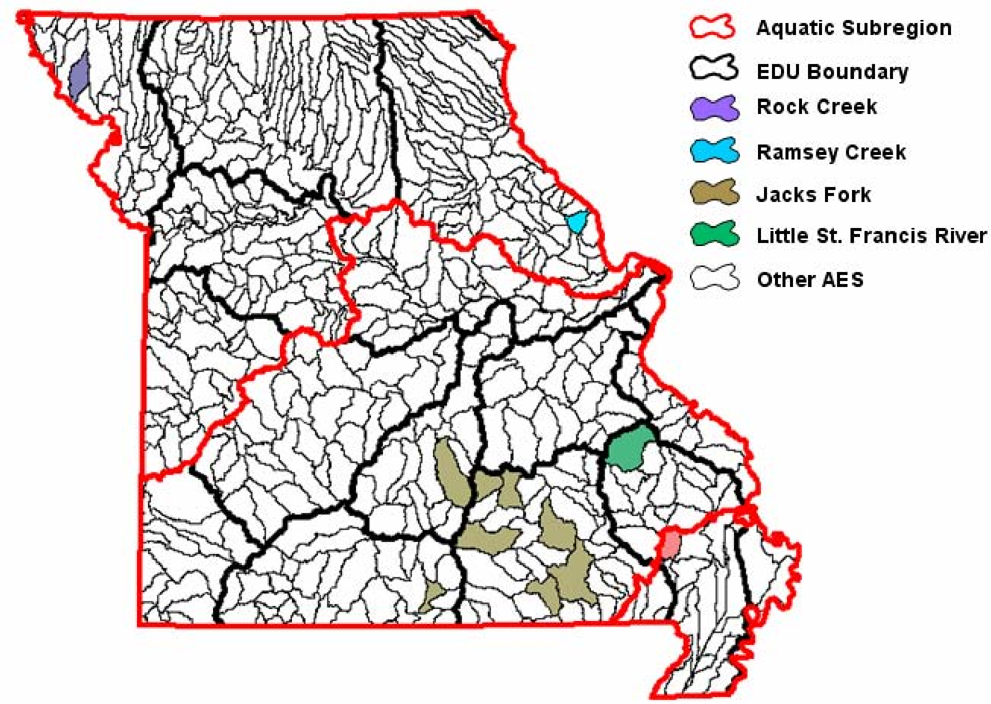


Figure 7.5. Map showing the individual Aquatic Ecological Systems (AESs) that have all stream size classes captured as an interconnected complex.

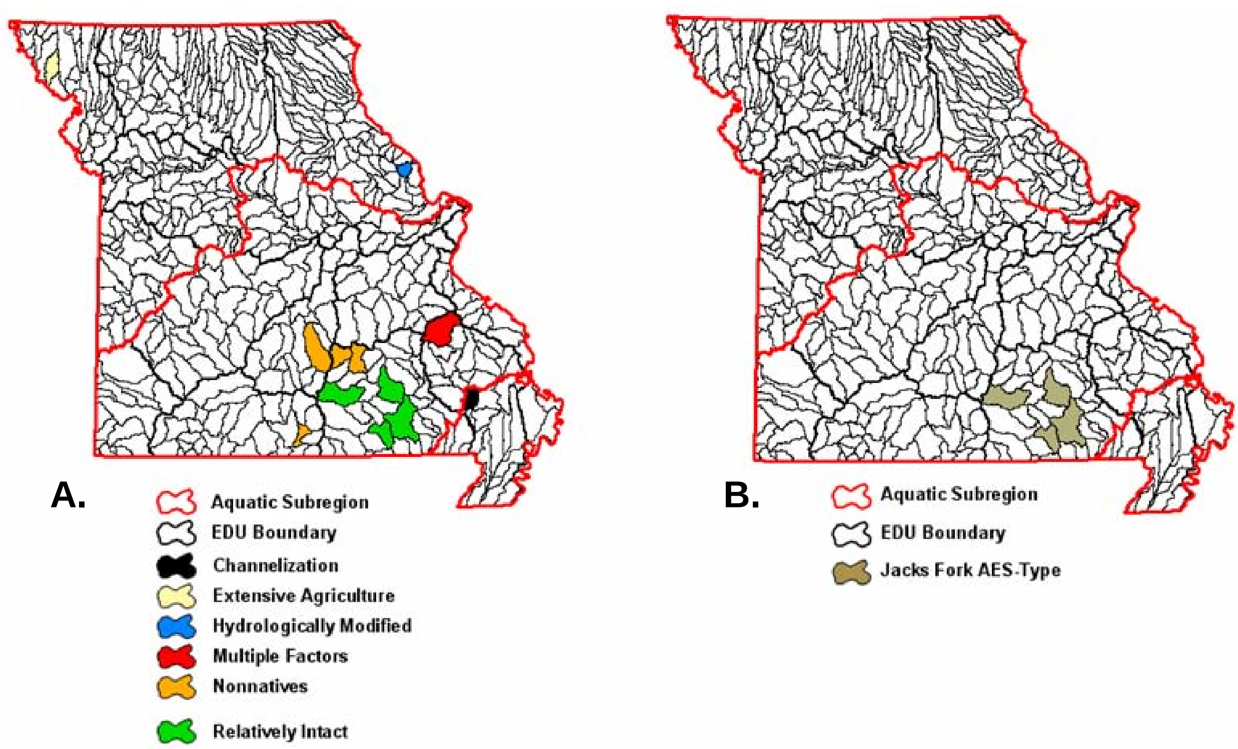


Figure 7.6. Maps showing the major human stressors affecting those AESs that have all stream sizes captured as an interconnected complex in Status 1 or 2 lands (map A.) and those remaining AESs that can be considered relatively intact (map B.).

7.4 Analysis of Biotic Elements

To assess the representation of fish, mussel, and crayfish species in status 1 or 2 lands we calculated the total length of stream in which each species was predicted to occur that flows through status 1 or 2 lands. We also calculated the number of AES polygons in which each species was represented in status 1 or 2 lands. The first calculation provides insight into which species have little or no suitable habitat currently represented in lands set aside for the long-term maintenance of biodiversity. The second calculation provides insight into how many distinct occurrences ("population subunits") of each species are represented in status 1 or 2 lands. For this second calculation we assume that each individual AES represents a distinct occurrence or population subunit for each species, which is not always a correct assumption especially for wide-ranging species. However, we believe that these statistics are important and do provide additional insight into how well each species is currently represented in the existing matrix of public lands. The above statistics were generated and reported from a statewide perspective and also within the ecosystem context provided by our Aquatic Subregions and Ecological Drainage Units.

Results for Analyses Based on Length

A total of 315 species of fish, mussels and crayfish have been collected within Missouri. Fourteen of these species are not native to Missouri and five of the native species occur in cave habitats (three fish and two crayfish), and were not included in our gap analyses. Of the remaining 296 native fish, mussel and crayfish species most have greater than 50 km of their predicted distribution within status 1 or 2 lands (Figure 7.7). In fact, anywhere from 40 to 70% of the species within each the three taxonomic groups have greater than 50 km of their distribution within status 1 or 2 lands (Figure 7.8). Appendices 7.1 and 7.2 provide the length and percent length contained within each gap management status category for all 315 species. A total of 45 (15%) native species (32 fish, 5 mussels, and 8 crayfish) are currently not represented in any status 1 or 2 lands (Table 7.8). The vast majority of these species (30, 67%) are state listed as rare, threatened, or endangered and ten are listed as globally rare, threatened, or endangered. Although these 45 species occur all across the state, the richness plot provided in Figure 7.9 shows that the highest concentration occurs within the Mississippi River, the Mississippi Alluvial Basin (MAB) Aquatic Subregion, and the Neosho EDU, located in southwestern Missouri. Also, within the MAB and the Neosho EDU, the highest concentration of these species occurs within the larger mainstem streams. The fact that most of the these species primarily inhabit larger rivers, represents a significant conservation challenge due to the large amount of land that must be managed and the cumulative effect of numerous human disturbances that are typically spread across the watersheds of these larger streams.

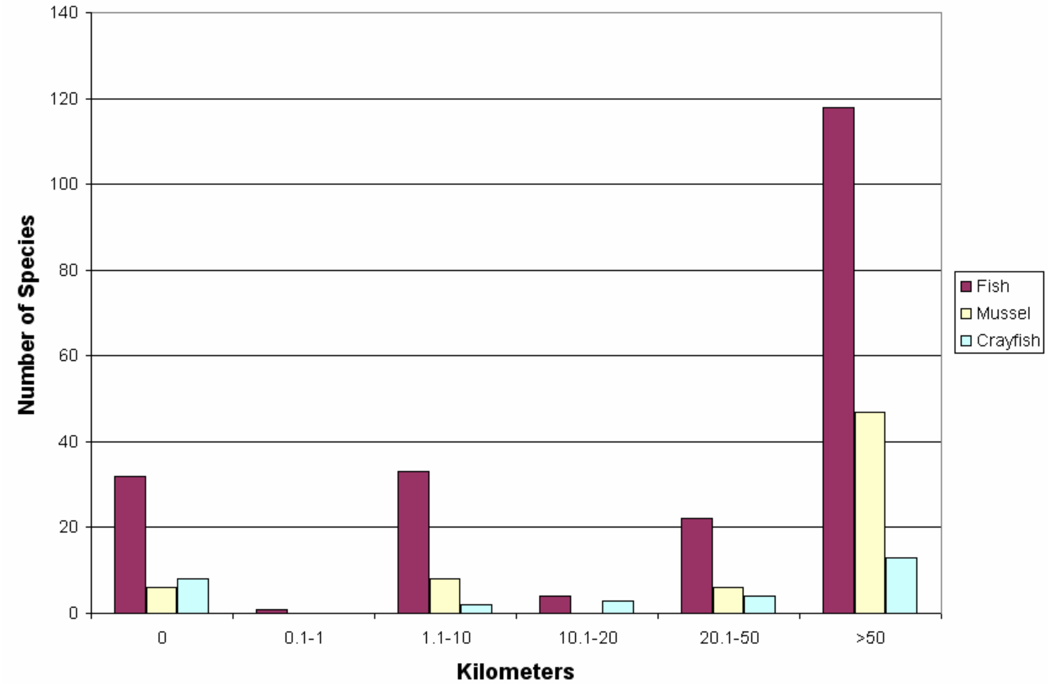


Figure 7.7. Bar chart showing the number of species, within each taxon, that occur within six levels of representation (status 1 or 2 lands) by length. For example, of the 210 fish species; 32 are not represented in any status 1 or 2 lands, 1 has between 0.1 and 1 km, 33 have between 1.1 and 10 km, 4 have between 10.1 and 20 km, 22 have between 20.1 and 50 km, and 118 have greater than 50 km represented in status 1 or 2 lands.

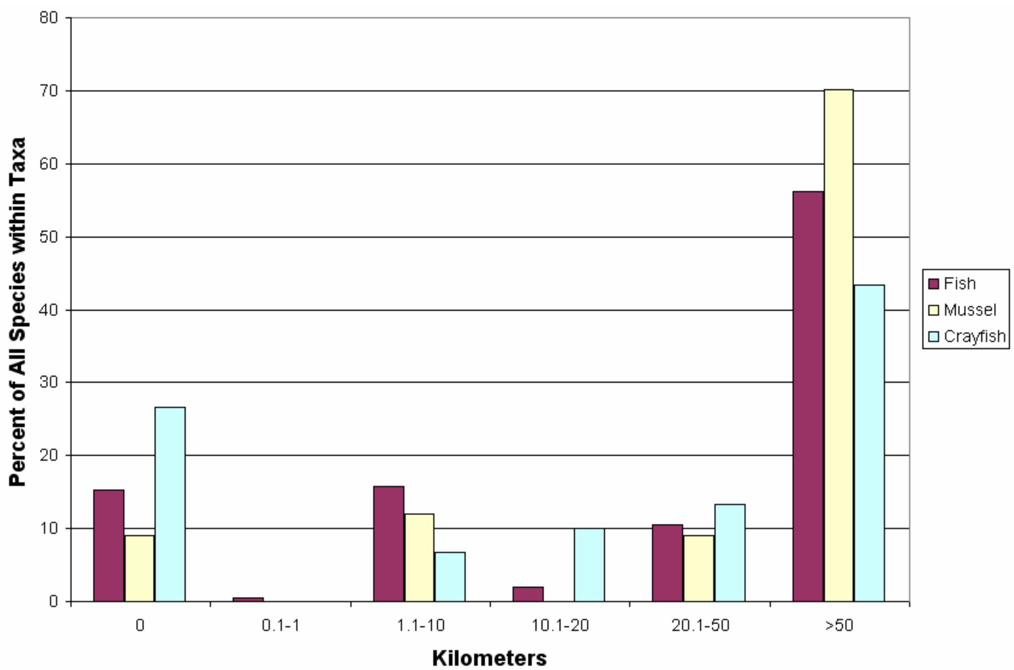


Figure 7.8. Bar chart showing the percentage of species, within each taxonomic, that fall within six levels of representation (status 1 or 2 lands) by length. For example, of the 30 crayfish species; 8 (27%) are not represented in any status 1 or 2 lands, 0 have between 0.1 and 1 km, 2 (7%) have between 1 and 10 km, 3 (10%) have between 10 and 20 km, 4 (13%) have between 20 and 50 km, and 13 (43%) have greater than 50 km represented in status 1 or 2 lands.

Table 7.8. List of the 45 species of fish, mussels and crayfish native to Missouri which are currently not represented in any status 1 or 2 lands. *Note: this list does not include five cave species (3 fish and 2 crayfish). Grank and Srank represent global and state conservation status ranks.*

Taxon	Common	Scientific	Grank	Srank
Fish	alligator gar	Atractosteus spatula	G3G4	SX
	bluntface shiner	Cyprinella camura	G5	S2S3
	brassy minnow	Hybognathus hankinsoni	G5	S3
	burbot	Lota lota	G5	S?
	central mudminnow	Umbra limi	G5	S1
	channel darter	Percina copelandi	G4	S3
	cypress minnow	Hybognathus hayi	G5	S1
	dollar sunfish	Lepomis marginatus	G5	SU
	golden topminnow	Fundulus chrysotus	G5	S1
	goldstripe darter	Etheostoma parvipinne	G4G5	S1
	harlequin darter	Etheostoma histrio	G5	S2
	inland silverside	Menidia beryllina	G5	S3
	ironcolor shiner	Notropis chalybaeus	G4	S1
	longnose darter	Percina nasuta	G3	S1
	mountain madtom	Noturus eleutherus	G4	S1S2
	Neosho madtom	Noturus placidus	G2	S2
	Niangua darter	Etheostoma nianguae	G2	S2
	northern pike	Esox lucius	G5	S4
	plains killifish	Fundulus zebrinus	G5	S2
	pumpkinseed	Lepomis gibbosus	G5	S?
	redfin darter	Etheostoma whipplei	G5	S1
	redspot chub	Nocomis asper	G4	S?
	Sabine shiner	Notropis sabinae	G3	S1
	silver lamprey	Ichthyomyzon unicuspis	G5	S?
	silverjaw minnow	Notropis buccatus	G5	S4
	spottail shiner	Notropis hudsonius	G5	S?
	striped mullet	Mugil cephalus	G5	SA
	swamp darter	Etheostoma fusiforme	G5	S1
	taillight shiner	Notropis maculatus	G5	S1
	threadfin shad	Dorosoma petenense	G5	S?
	yellow bass	Morone mississippiensis	G5	S?
	yellow perch	Perca flavescens	G5	S?
Mussel	fat pocketbook	Potamilus capax	G1	S1
	hickorynut	Obovaria olivaria	G4	S2S3
	Higgins eye	Lampsilis higginsii	G1	SA
	southern hickorynut	Obovaria jacksoniana	G1G2	S1
	Texas lilliput	Toxolasma texasensis	G4	S3
Crayfish	Cajun dwarf crayfish	Cambarellus puer	G4G5	S3?
	digger crayfish	Fallicambarus fodiens	G5	S2S3
	Mammoth Spring crayfish	Orconectes marchandi	G2	S1S2
	Meek's crayfish	Orconectes meeki	G4	S1
	Neosho midget crayfish	Orconectes macrus	G4	S3?
	shrimp crayfish	Orconectes lancifer	G5	S1S2
	white river crayfish	Procambarus acutus	G5	S?
	Williams' crayfish	Orconectes williamsi	G2	S1?

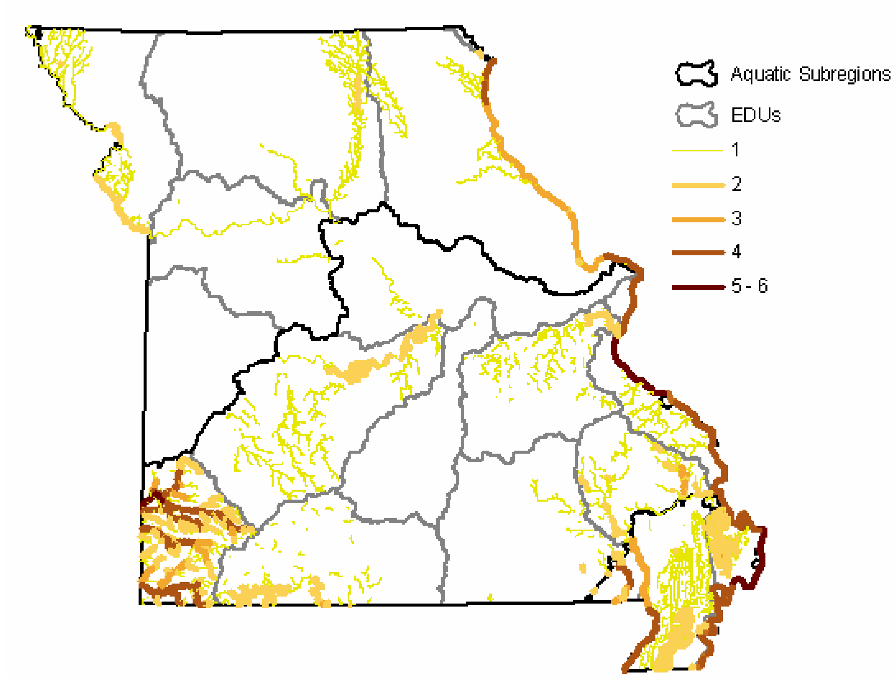


Figure 7.9. Map of species richness for the 45 native fish, mussel, and crayfish species that are currently not represented in any status 1 or 2 conservation lands.

When we examined the representation of species by Aquatic Subregion, we found that the Ozarks has the highest percentage of species represented in status 1 or 2 lands followed by the MAB and the lowest percentage occurring in the Central Plains (Figure 7.10). Specifically, there are 278 native fish, mussel, and crayfish species that occur within the Ozarks, of which 52 (19%) are not represented within the status 1 or 2 lands that occur within this Subregion. Within the MAB there are 163 native species and 69 (42%) do not occur in the status 1 or 2 lands that occur in this Subregion. Finally, of the 178 native species that occur within the Central Plains Aquatic Subregion, 90(51%) are not currently represented in any of the status 1 or 2 lands that occur within this Subregion.

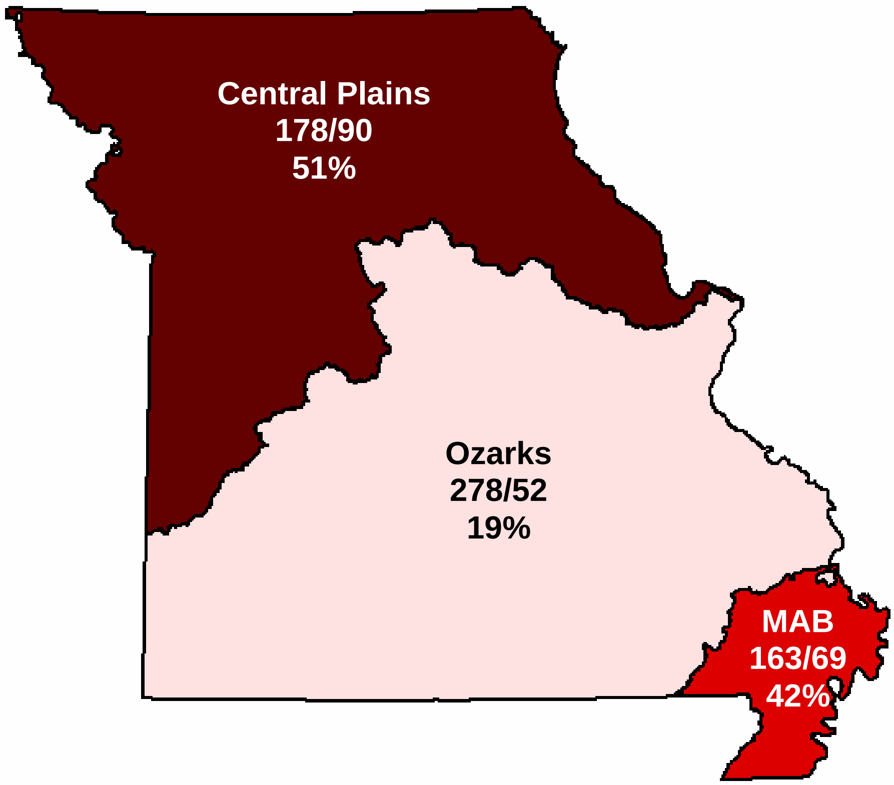


Figure 7.10. Map showing the total number of native fish, mussel, and crayfish species that occur within each Aquatic Subregion followed by the number and percent of native species not represented within the status 1 or 2 lands that occur within each respective subregion.

Representation of native species within each EDU shows a high degree of variation (Table 7.9; Figure 7.11). None of the EDUs have all of the native species occurring within their boundaries currently represented in status 1 or 2 lands. Within the Central Plains only 34% of the native species that occur within the Grand/Chariton EDU were not represented within the status 1 or 2 lands that occur within this EDU, compared with 78% within the Blackwater/Lamine EDU. Within the Ozarks four EDUs had 25% or less of the native species not represented in status 1 or 2 lands. However, the Osage (79%), Apple/Joachim (83%), and the Neosho (84%) EDUs all had less than 75% of their native species represented in status 1 or 2 lands. Within the MAB Aquatic Subregion there is fairly good representation within the St. Francis/Little EDU with only 25% of the native species not represented in status 1 or 2 lands. However, the Black/Cache and St. John's Bayou EDUs both have more than 50% of the native species not represented. The high degree of variation in these EDU-level statistics further illustrates the spatially biased and highly fragmented nature of the public lands in Missouri.

Table 7.9. Statistics illustrating how well native fish, mussel, and crayfish species are represented in status 1 or 2 lands within each Ecological Drainage Unit (EDU). Table shows the total number of native species that occur within each EDU and the number and percent of these species that are not currently represented within the status 1 or 2 lands that occur within each EDU.

Aquatic Subregion	Ecological Drainage Unit	Total Native Species	Species Not Represented in Status 1 or 2	Percent Not Represented
Central Plains	Blackwater/Lamine	122	95	78
	Cuivre/Salt	158	87	55
	Grand/Chariton	92	30	33
	Nishnabotna/Platte	91	60	66
	Osage/South Grand	110	40	36
Ozarks	Apple/Joachim	142	118	83
	Black/Current	187	32	17
	Gasconade	150	35	23
	Meramec	172	43	25
	Moreau/Loutre	137	34	25
	Neosho	132	111	84
	Osage	158	125	79
	Upper St. Francis/Castor	181	63	35
	White	139	50	36
Mississippi Alluvial Basin	Black/Cache	109	60	55
	St. Francis/Little	126	31	25
	St. Johns Bayou	135	87	64

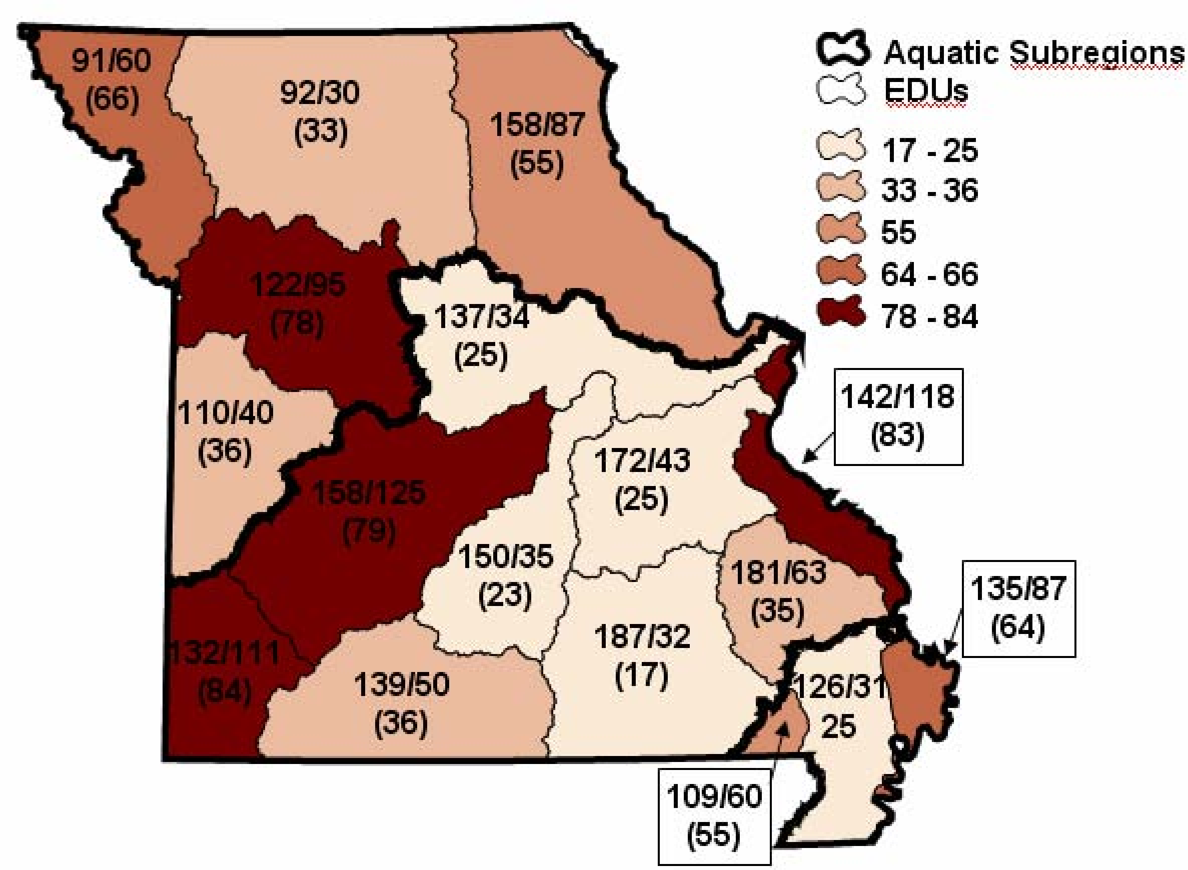


Figure 7.11. Map showing the total number of native fish, mussel, and crayfish species within each Ecological Drainage Unit (EDU) and the corresponding number and percent of those species that are NOT represented within the status 1 or 2 lands that occur within each respective EDU. Graduated colors represent the percent of species not represented in status 1 or 2 lands.

Results for Analyses Based on Distinct Occurrences

From a statewide perspective, most of the 296 native fish, mussel and crayfish species (227, 77%) have multiple distinct occurrences or populations subunits represented in status 1 or 2 lands (Figure 7.12). In fact, a high percentage of the species within each of the three taxonomic groups (23-49%) have greater than ten distinct population subunits represented in status 1 or 2 lands (Figure 7.13). Appendix 7.3 provides the number of distinct occurrences represented in status 1 or 2 lands for all of the species included in our analyses. A total of 69 (23%) native species (50 fish, 8 mussels, and 11 crayfish) have less than two distinct occurrences represented in status 1 or 2 lands (Table 7.10). The vast majority of these species (49, 71%) are state listed as rare, threatened, or endangered and 21 are listed as globally rare, threatened, or endangered.

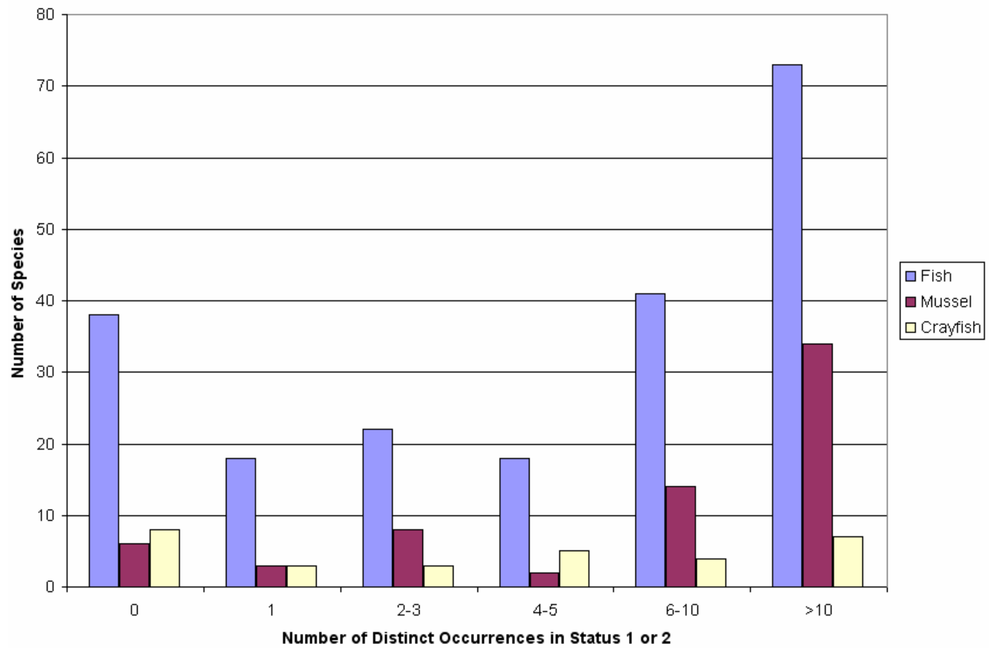


Figure 7.12. Bar chart showing the number of species, within each taxon, that fall within six levels of representation (status 1 or 2 lands) by distinct occurrence. For example, of the 210 fish species; 32 are not represented in any status 1 or 2 lands, 18 have one distinct occurrence, 22 have either 2 or 3 distinct occurrences, 18 have either 4 or 5 distinct occurrences, 41 have between 6 and 10 distinct occurrences, and 73 have greater than 10 distinct occurrences (population subunits) represented in status 1 or 2 lands.

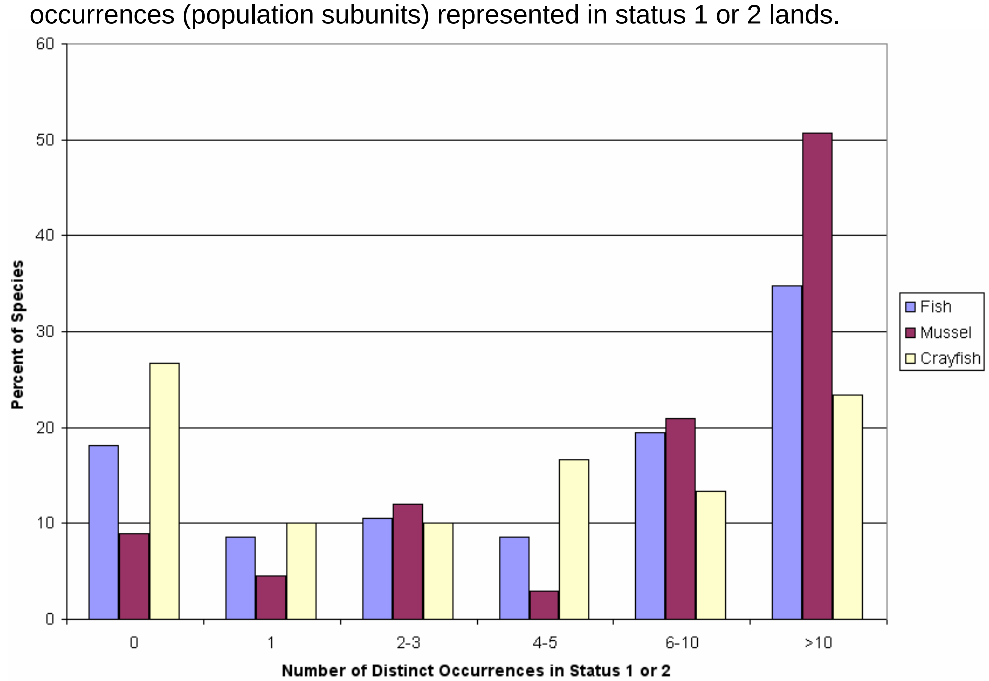


Figure 7.13. Bar chart showing the percent of species, within each taxon, that fall within six levels of representation (status 1 or 2 lands) by distinct occurrence. For example, of the 210 fish species; 18% are not represented in any status 1 or 2 lands, 9% have one distinct occurrence, 10% have either 2 or 3 distinct occurrences, 9% have either 4 or 5 distinct occurrences, 20% have between 6 and 10 distinct occurrences, and 35% have greater than 10 distinct occurrences (population subunits) represented in status 1 or 2 lands.

Table 7.10. List of the 69 species of fish, mussels and crayfish native to Missouri with less than 2 distinct occurrences represented in any status 1 or 2 lands. *Note: this list does not include five cave species (3 fish and 2 crayfish). Grank and Srank represent global and state conservation status ranks.*

Taxon	Common Name	Scientific Name	Grank	Srank
Fish	Alabama shad	Alosa alabamae	G3	S2
	alligator gar	Atractosteus spatula	G3G4	SX
	Arkansas darter	Etheostoma cragini	G3	S3
	blue catfish	Ictalurus furcatus	G5	SU
	bluntface shiner	Cyprinella camura	G5	S2S3
	brassy minnow	Hybognathus hankinsoni	G5	S3
	brindled madtom	Noturus miurus	G5	S?
	brown bullhead	Ameiurus nebulosus	G5	S3?
	burbot	Lota lota	G5	S?
	cardinal shiner	Luxilus cardinalis	G4	S?
	central mudminnow	Umbra limi	G5	S1
	channel darter	Percina copelandi	G4	S3
	channel shiner	Notropis wickliffi	G5	S?
	cypress minnow	Hybognathus hayi	G5	S1
	dollar sunfish	Lepomis marginatus	G5	SU
	flathead chub	Platygobio gracilis	G5	S1
	golden topminnow	Fundulus chrysotus	G5	S1
	goldstripe darter	Etheostoma parvipinne	G4G5	S1
	harlequin darter	Etheostoma histrio	G5	S2
	inland silverside	Menidia beryllina	G5	S3
	ironcolor shiner	Notropis chalybaeus	G4	S1
	lake sturgeon	Acipenser fulvescens	G3	S1
	least darter	Etheostoma microperca	G5	S2
	longnose darter	Percina nasuta	G3	S1
	mountain madtom	Noturus eleutherus	G4	S1S2
	Neosho madtom	Noturus placidus	G2	S2
	Niangua darter	Etheostoma nianguae	G2	S2
	northern pike	Esox lucius	G5	S4
	pallid sturgeon	Scaphirhynchus albus	G1	S1
	plains killifish	Fundulus zebrinus	G5	S2
	pumpkinseed	Lepomis gibbosus	G5	S?
	redfin darter	Etheostoma whipplei	G5	S1
	redspot chub	Nocomis asper	G4	S?
	Sabine shiner	Notropis sabinae	G3	S1
	shoal chub	Macrhybopsis hyostoma	G5	S?
	sicklefin chub	Macrhybopsis meeki	G3	S3
	silver lamprey	Ichthyomyzon unicuspis	G5	S?
	silverband shiner	Notropis shumardi	G5	S?
	silverjaw minnow	Notropis buccatus	G5	S4
	spottail shiner	Notropis hudsonius	G5	S?
	stargazing darter	Percina uranidea	G3	S2
	starhead topminnow	Fundulus dispar	G4	S2
	striped mullet	Mugil cephalus	G5	SA
	sturgeon chub	Macrhybopsis gelida	G3	S3

Table 7.10. Continued.

Taxon	Common Name	Scientific Name	Grank	Srank
	swamp darter	Etheostoma fusiforme	G5	S1
	taillight shiner	Notropis maculatus	G5	S1
	threadfin shad	Dorosoma petenense	G5	S?
	Topeka shiner	Notropis topeka	G2	S1
	yellow bass	Morone mississippiensis	G5	S?
	yellow perch	Perca flavescens	G5	S?
Mussel	cylindrical papershell	Anodontoides ferussacianus	G5	S1?
	fat pocketbook	Potamilus capax	G1	S1
	hickorynut	Obovaria olivaria	G4	S2S3
	Higgins eye	Lampsilis higginsii	G1	SA
	Neosho mucket	Lampsilis rafinesqueana	G2	S2
	salamander mussel	Simpsonaias ambigua	G3	S1?
	southern hickorynut	Obovaria jacksoniana	G1G2	S1
	Texas lilliput	Toxolasma texasensis	G4	S3
Crayfish	belted crayfish	Orconectes harrisonii	G3	S3
	Cajun dwarf crayfish	Cambarellus puer	G4G5	S3?
	digger crayfish	Fallicambarus fodiens	G5	S2S3
	Mammoth Spring crayfish	Orconectes marchandi	G2	S1S2
	Meek's crayfish	Orconectes meeki	G4	S1
	Neosho midget crayfish	Orconectes macrus	G4	S3?
	shield crayfish	Faxonella clypeata	G5	S2S3
	shrimp crayfish	Orconectes lancifer	G5	S1S2
	vernal crayfish	Procambarus viaeviridis	G5	S3?
	white river crayfish	Procambarus acutus	G5	S?
	Williams' crayfish	Orconectes williamsi	G2	S1?

From the perspective of our Aquatic Subregions we find a dramatic drop in the percentage of species that have multiple distinct occurrences represented in status 1 or 2 lands. The Ozarks has the highest percentage followed by the MAB and the lowest percentage occurring in the Central Plains (Figure 7.14). Specifically, there are 278 native fish, mussel, and crayfish species that occur within the Ozarks, of which 81 (29%) have less than two distinct occurrences represented within the status 1 or 2 lands that occur within this Subregion. Within the MAB there are 163 native species and 76 (47%) have less than two distinct occurrences within the status 1 or 2 lands that occur in this Subregion. Finally, of the 178 native species that occur within the Central Plains Aquatic Subregion, 109(61%) have less than two distinct occurrences represented in the status 1 or 2 lands that occur within this Subregion.

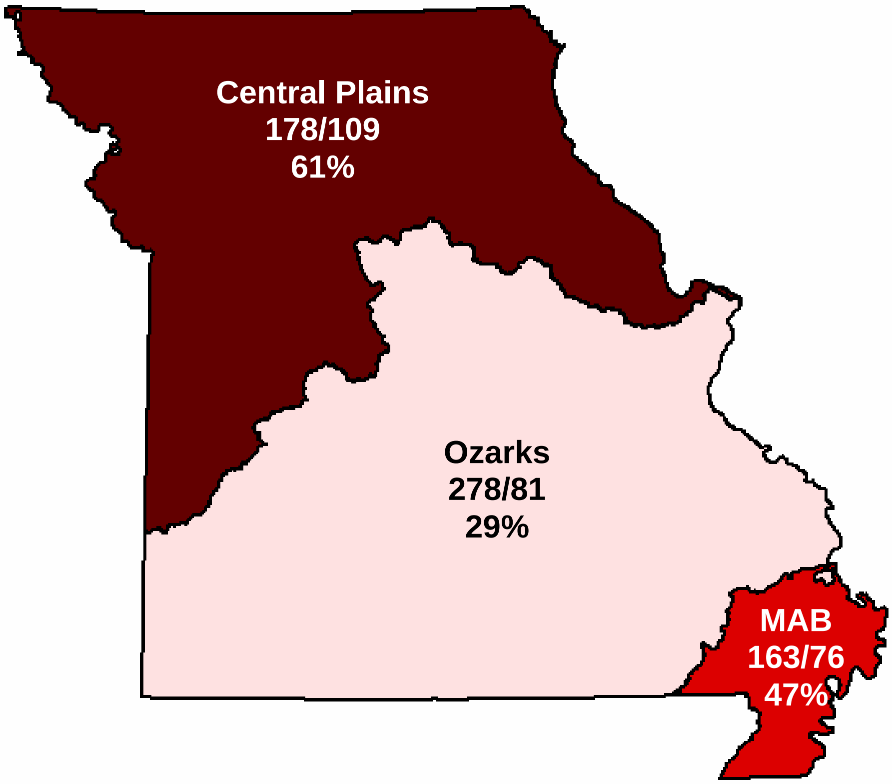


Figure 7.14. Map showing the total number of native fish, mussel, and crayfish species that occur within each Aquatic Subregion followed by the number and percent of native species that have less than two distinct occurrences represented within the status 1 or 2 lands that occur within each respective subregion.

Similar to what was found for the analyses by length, results of the analyses examining distinct occurrences by EDU revealed a high degree of variation (Table 7.11; Figure 7.15). None of the EDUs had all of the native species, occurring within their boundaries, represented more than once within status 1 or 2 lands. Within the Central Plains 62% of the native species that occur within the Grand/Chariton EDU had multiple population subunits represented the status 1 or 2 lands that occur within this EDU, however none of the species within the Nishnabotna/Platte EDU were represented more than once. Within the Ozarks 73% of the native species that occur within the Black/Current EDU had multiple population subunits represented status 1 or 2 lands, however, none of the species within the Neosho EDU were represented more than once. Within the MAB Aquatic Subregion there is fairly good representation within the St. Francis/Little EDU with 69% of the native species represented more than once within status 1 or 2 lands. However, none of the native species within the Black/Cache and St. John's Bayou EDUs were represented more than once.

Table 7.11. Statistics pertaining to the redundancy of representation for native fish, mussel, and crayfish species within each Ecological Drainage Unit (EDU). Table shows the total number of native species that occur within each EDU and the number and percent of those species that have less than two distinct occurrences within the status 1 or 2 lands that occur within each EDU.

Subregion	Ecological Drainage Unit	Total Native Species	Species with less than 2 distinct occurrences in Status 1 or 2	Percent with less than 2 distinct occurrences in Status 1 or 2
Central Plains	Blackwater/Lamine	122	102	84
	Cuivre/Salt	158	133	84
	Grand/Chariton	92	35	38
	Nishnabotna/Platte	91	91	100
	Osage/South Grand	110	81	74
Ozarks	Apple/Joachim	142	125	88
	Black/Current	187	50	27
	Gasconade	150	50	33
	Meramec	172	102	59
	Moreau/Loutre	137	90	66
	Neosho	132	132	100
	Osage	158	140	89
	Upper St. Francis/Castor	181	84	46
	White	139	56	40
Mississippi Alluvial Basin	Black/Cache	109	109	100
	St. Francis/Little	126	39	31
	St. Johns Bayou	135	135	100

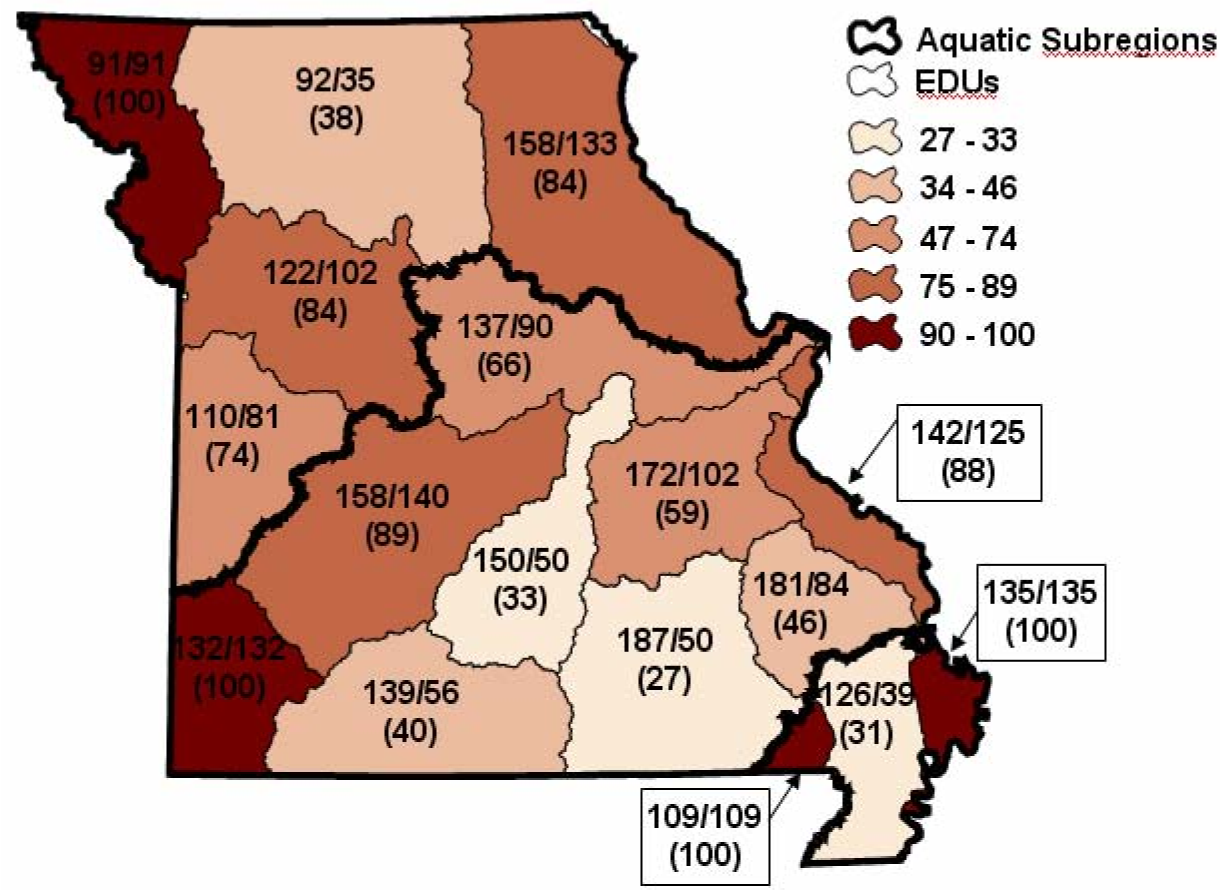


Figure 7.15. Map showing the total number of native fish, mussel, and crayfish species within each Ecological Drainage Unit (EDU) and the corresponding number and percent of those species that have less than two distinct occurrences represented in status 1 or 2 lands that occur within each respective EDU. Graduated colors represent the percent of species having less than two distinct occurrences.

7.5. Discussion and Limitations

The results of the analyses performed in this chapter are based on models and mapping data that are certainly not without error. Please refer to chapters 3-6 and the metadata provided for each of the geospatial datasets used in these analyses for a thorough discussion of the methods used to create these data and their respective limitations. The results presented in this chapter should be interpreted from a general perspective since the data are suited to coarse-scale assessments that illustrate general patterns of representation of abiotic and biotic elements of biodiversity within the existing matrix of public lands in Missouri.

Just over 5% (9,373 km) of the 174,063 total kilometers of stream within Missouri are contained within the existing matrix of public lands. The vast majority of these streams (85.2%) flow through lands classified as GAP management status 3. Less than 1%

(1,342 km) flow through lands that are primarily managed for the long-term maintenance of biodiversity (i.e., management status 1 or 2), and 88% of these stream miles occur within the Ozarks.

From a statewide perspective, a relatively high percentage of the Valley Segment Types (VSTs) (77%) and riverine species (85%) are represented in status 1 or 2 lands. Most of the 17 VSTs that are not represented fall within the smaller headwater or creek size classes, which is not surprising considering the higher diversity of VSTs for these smaller streams. Most of the 45 species not represented in status 1 or 2 lands are fishes (32). The highest concentration of these species occurs within the Mississippi River, the Mississippi Alluvial Basin (MAB) Aquatic Subregion, and the Neosho Ecological Drainage Unit (EDU), located in southwestern Missouri. The fact that most of these species are state listed species that can be considered either large-river species or local endemics, presents a significant challenge to stream resource managers. Conserving large rivers is obviously difficult due to the enormous land area that must be managed, but the diversity and cumulative nature of the human disturbances adds to the complexity of the management efforts. Local endemics present a management challenge because very little is known about the life-history requirements of these species and in Missouri they occur as widely scattered populations mainly across the Ozark Aquatic Subregion.

The statewide gap analyses for the Aquatic Ecological System Types (AES-Types) clearly illustrated the fragmented spatial distribution of public lands in Missouri. Only 19 of the 542 individual AESs had all three stream size classes represented in status 1 or 2 lands and only 12 of these had all size classes represented as an interconnected matrix. An assessment of human disturbances, within the 12 AESs that met the above criteria, revealed that only 4 could be considered relatively undisturbed or ecologically intact. The other 8 units were significantly altered as a result of intensive agriculture, channelization, nonnative species, or hydrologic modification due to a large impoundment. The statewide analyses for the AES-Types also revealed a high degree of redundancy in representation. Thirteen of the 19 AESs, that had all stream sizes represented, were the same type of riverine ecosystem (Jack's Fork AES-Type). The final 4 AESs, which met all of the above criteria, were also all of the same ecosystem type and all were located within the same EDU, which represents an extreme example of redundancy in representation. This fragmented nature of public ownership is probably one of the most pressing conservation issues that must be addressed in the future. When you consider that many riverine species require a range of stream sizes to meet all of their life history requirements (Schlosser 1995), it becomes evident that in Missouri greater attention must be paid to the spatial arrangement of future conservation lands or private land conservation measures.

The representation of both abiotic and biotic elements dropped considerably when the analyses were conducted separately for each Aquatic Subregion and EDU. These analyses collectively reveal the enormity of the challenge we face when it comes to the long-term conservation of freshwater biodiversity in Missouri. More specifically, if you agree with the contentions of the stream resource managers in Missouri (see Chapter 8) that measures should be taken to holistically conserve each EDU in the state, then our EDU-level analyses provide direct insight into how well this conservation objective is being met. The clearest perspective on how well we achieving this objective is provided by a specifically examining the “best case scenario” in terms representation of abiotic and biotic elements, which is the Black/Current EDU within the Ozarks. Twenty of the 54 VSTs (37%) and 32 of the 187 native species (17%) that occur within this EDU are not represented in status 1 or 2 lands. Furthermore, only one of the 9 AES-Types that occur within this EDU have all stream sizes represented either separately or as an interconnected matrix. Again, these statistics represent the best case scenario, whereas in many other EDUs, like the Neosho or St. John’s Bayou, we are essentially starting with a “clean slate” in terms of representation. These results clearly indicate that the existing public lands in Missouri do not even come close to holistically representing the full spectrum of freshwater biodiversity, especially at higher levels of ecological organization.

Since most of Missouri and its stream resources are within private ownership, successful conservation of freshwater biodiversity will require creative partnerships between resource agencies and private land owners. The many federal and state conservation incentive programs that are currently used as management tools are certainly a step in the right direction. However, we believe the results our gap analyses illustrate the need for a more strategic approach to where these conservation measures applied on the landscape. Randomly applying the conservation measures across the landscape will likely not provide the same level of benefits as would efforts directed at restoring and protecting key locations across the riverscape that represent the diversity of freshwater ecosystems in Missouri. The data we have developed for the Missouri Aquatic GAP Project are perfectly suited to develop such strategies as will be illustrated in the following chapter.

CHAPTER 8

Developing a Conservation Plan for Conserving Missouri's Freshwater Biodiversity

Failing to plan is planning to fail. - Alan Lakein

8.1 Background

In fall 2001, federal legislation established a new State Wildlife Grants (SWG) program, in order to provide funds to state wildlife agencies for the conservation of fish and wildlife species, including nongame species. For states to continue receiving federal funds through the SWG program, Congress charged each state and territory with developing a statewide Comprehensive Wildlife Conservation Strategy (CWCS). In Missouri, the Conservation Department (MDC) is responsible for developing the CWCS. MoRAP worked with MDC to develop customized GIS projects that would assist in the development of a statewide plan for conserving freshwater biodiversity. These customized GIS projects included all of the data compiled or created for the Missouri Aquatic GAP Project, as well as other pertinent geospatial data. At the same time, MDC developed customized GIS projects for developing a statewide plan for conserving terrestrial biodiversity. Interim results of these two plans were merged into a single CWCS for the state. This chapter covers the methods and results of the statewide plan for conserving freshwater biodiversity.

After the customized GIS projects were developed, a team of aquatic resource professionals from around Missouri was assembled. The objective of this team was to address each of the basic components of conservation planning discussed in Chapter 2 of this report.

The team formulated the following goal:

Ensure the long-term persistence of native aquatic plant and animal communities, by conserving the conditions and processes that sustain them, so people may benefit from their values in the future.

The team then put together a list of principles, theories, and assumptions that must be considered in order to achieve this goal. Most were similar to those presented in Chapter 2 and related mainly to basic principles of stream ecology, landscape ecology, and conservation biology. However, some reflected the personal experiences of team members and the challenges they face when conserving natural resources in regions with limited public land holdings. For instance, one of the assumptions identified by the team was: "Success will often hinge upon the participation of local stakeholders, which will often be private landowners." In fact, the importance of private lands management to aquatic biodiversity conservation was a topic that permeated throughout the initial meetings of the team.

The MoRAP aquatic ecological classification hierarchy (see Chapter 3) was adopted as the geographic framework (i.e., Planning Regions and Asssessment Units) for developing the conservation plan. From this classification hierarchy they selected AES-Types and VSTs as abiotic conservation targets. They also agreed that, in order to fully address biotic targets, a list of target species (fish, mussel, and crayfish) should be developed for each EDU. These lists were developed and they represent species of conservation concern (i.e, global ranks: G1-G3 and state ranks: S1-S3), endemic species, and focal or characteristic species (e.g., top predators, dominant prey species, unique ecological role, etc.). It was also agreed that all biological statistics would be based on the predicted distributions developed for the Missouri Aquatic GAP Project (see Chapter 4), but that actual collection data would also be used when it was deemed necessary during the planning process.

8.2 The Conservation Strategy

Next the team crafted a general conservation strategy that would be used to identify and map a statewide portfolio of Conservation Opportunity Areas (COAs) that collectively and holistically represent all of the distinct riverine ecosystems within Missouri and multiple populations of all fish, mussel, and crayfish species. The reasoning behind each component of this strategy is best illustrated by discussing what conservation objectives the team hoped to achieve with each component. These reasons are provided in Box 8.1, below.

Basic Elements of the Conservation Strategy:

- We must develop **separate conservation plans for each EDU**;
- whenever possible, represent **two** distinct spatial occurrences/populations of each target species within each EDU;
- represent at least **one example of each AES-Type** within each EDU;
- within each selected AES, **represent at least 1 km of the dominant VSTs for each size class** (headwater, creek, small river, and large river) as an interconnected complex; and
- represent a least **three separate headwater VSTs** within each of the Conservation Opportunity Areas.

Box 8.1. Explanation of what we were attempting to achieve with each component of the general conservation strategy that was used to select conservation opportunity areas for protecting freshwater biodiversity throughout Missouri.

By attempting to conserve every EDU

- Provide a holistic ecosystem approach to biodiversity conservation, since each EDU represents an interacting biophysical system
- Represent all of the characteristic species and species of concern within the broader Aquatic Subregion and the entire state, since no single EDU contains the full range of species found within the upper levels of the classification hierarchy
- Represent multiple distinct spatial occurrences ("populations") or phylogenies for large-river or wide-ranging species (e.g., sturgeon, catfish, paddlefish), which, from a population standpoint, can only be captured once in any given EDU

By attempting to conserve two distinct occurrences of each Target Species within each EDU

- Provide redundancy in the representation of those species that collectively determine the distinctive biological composition of each EDU in order to provide a safeguard for the long term persistence of these species

By attempting to conserve an individual example of each AES-Type within each EDU

- Represent a wide spectrum of the diversity of macrohabitats (distinct watershed types) within each EDU
- Account for successional pathways and safeguard against long-term changes in environmental conditions caused by factors like Global Climate Change.
    - For instance, gross climatic or land use changes may make conditions in one AES-Type unsuitable for a certain species, but at the same time make conditions in another AES-Type more favorable for that species
- Represent multiple distinct spatial occurrences ("populations") for species with moderate (e.g., bass or sucker species) and limited dispersal capabilities (e.g., darters, sculpins, certain minnow species, most crayfish and mussels)
- Account for metapopulation dynamics (source/sink dynamics)

By attempting to conserve the dominant VSTs for each size class within a single AES

- Represent the dominant physicochemical conditions within each AES, which we assume represent the environmental conditions to which most species in the assemblage have evolved adaptations for maximizing growth, reproduction and survival (*sensu* Southwood 1977)
- Represent a wide spectrum of the diversity of mesohabitats (i.e., stream types) within each EDU since the dominant stream types vary among AES-Types
- Promote an ecosystem approach to biodiversity conservation by representing VSTs within a single watershed
- Account for metapopulation dynamics (source/sink dynamics)

Box 8.1. Continued.

By attempting to conserve an interconnected complex of dominant VSTs

- Account for seasonal and ontogenetic changes in habitat use or changes in habitat use brought about by disturbance (floods and droughts)
  - For instance, during periods of severe drought many headwater species may have to seek refuge in larger streams in order to find any form of suitable habitat due to the lack of water or flow in the headwaters
- Account for metapopulation dynamics (source/sink dynamics)
- Further promote an ecosystem approach to conservation by conserving an interconnected/interacting system

By attempting to conserve at least 3 headwater VSTs within each COA

- Represent multiple distinct spatial occurrences ("populations") for headwater species with limited dispersal capabilities (e.g., darters, sculpins, certain minnow species, most crayfish and mussels)
- Represent multiple high-quality examples of key reproductive or nursery habitats for many species

By attempting to conserve at least a 1 km of each priority VST

- Represent a wide spectrum of the diversity of Habitat Types (e.g., riffles, pools, runs, backwaters, etc.) within each VST and ensure connectivity of these habitats
- Account for seasonal and ontogenetic changes in local habitat use or changes in habitat use brought about by disturbance (e.g., floods and droughts)
  - For instance, many species require different habitats for foraging (deep habitats with high amounts of cover), reproduction (high gradient riffles), over-wintering (extremely deep habitats with flow refugia or thermally stable habitats like spring branches), or disturbance avoidance (deep or shallow habitats with flow refugia).
- Account for metapopulation dynamics (source/sink dynamics)
- Again, further promote an ecosystem approach to biodiversity conservation by representing an interacting system of Habitat Types

The team then established quantitative and qualitative assessment criteria for making relative comparisons among the assessment units. Since the assessment was conducted at two spatial grains (AES and VST), there exist two different assessment units with assessment criteria developed separately for each.

<u>AES level criteria (listed in order of importance)</u>

- Highest predicted richness of target species
- Lowest Human Stressor Index value and also qualitatively examine threats posed by individual human stressors
- Highest percentage of public ownership
- Overlap with existing conservation initiatives

- Ability to achieve connectivity among dominant VSTs across size classes
- When necessary, incorporate professional knowledge of opportunities, constraints, or human stressors not captured within the GIS projects to guide the above decisions.

VST level criteria (listed in order of importance)

- If possible, select a complex of valley segments that contains known viable populations of species of special concern.
- If possible, select the highest quality complex of valley segments by qualitatively evaluating the relative local and watershed conditions using the full breadth of available human stressor data.
- If possible, select a complex of valley segments that is already within the existing matrix of public lands.
- If possible, select a complex valley segments that overlaps with existing conservation initiatives or where local support for conservation is high.
- When necessary, incorporate professional knowledge of opportunities, constraints, or human stressors not captured within the GIS projects to guide above decisions.

The conservation strategy and assessment boils down to a five-step process:

1) Use the AES selection criteria to identify one priority AES for each AES-Type within the EDU.
2) Within each priority AES, use the VST selection criteria, to identify a priority complex of the dominant VSTs.
3) For each complex of VSTs create a map of the localized subdrainage (termed "Conservation Opportunity Area" (COA)) that specifically contains the entire interconnected complex.
4) Evaluate the capture of target species.
5) If necessary, select additional COAs to capture underrepresented target species.

8.3. Results

The team then used the conservation strategy and assessment process to develop a conservation plan for the Meramec EDU, which served as the initial pilot area for the statewide conservation plan. By using the above process all elements of the conservation strategy were met with 11 COAs (Figure 8.1). With the initial assessment process and selection criteria, which focus on abiotic targets (AESs and VSTs), 10 separate COAs were selected. These 10 areas represent the broad diversity of watershed and stream types that occur throughout the Meramec EDU. Within this initial set of 10 COAs, all but five of the 103 target species were captured (Appendix 8.1). The distribution of all five of these species overlapped within the same general area of the EDU, near the confluence of the Meramec and Dry Fork Rivers. Consequently, all five of these species were captured by adding a single COA (Dry Fork/Upper Meramec) (see Figure 8.1).

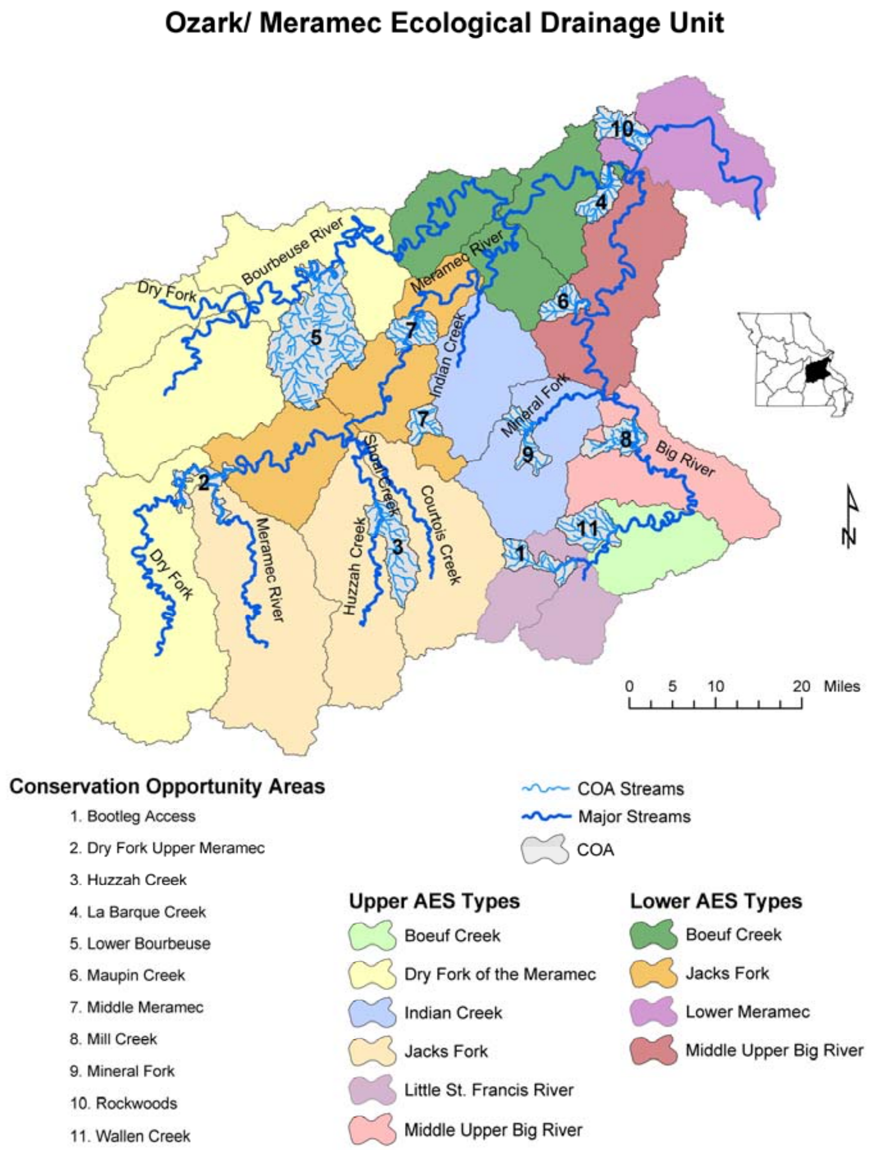


Figure 8.1. Map of 11 Conservation Opportunity Areas, within the Ozark/Meramec EDU, that were selected to meet all elements of the basic conservation strategy developed for the freshwater biodiversity conservation planning process in Missouri. The figure also shows the Aquatic Ecological System Types for context. Lower and Upper types differ in terms of their position within the larger drainage network. Specifically, a "Lower AES Type" contains streams classified as Large River and associated headwater and creek tributaries, while Upper types contain streams classified as Small River and these smaller tributaries.

The final set of priority valley segments, within the 11 COAs, constitutes 299 km of stream. This represents 2.8% of the total length of stream within the Meramec EDU. The COAs themselves represent an overall area of 552 km², which is just 5% of the nearly 10,360 km² contained within the EDU. Obviously, efforts to conserve the overall ecological integrity of the Meramec EDU cannot be strictly limited to the land area and stream segments within these COAs. In some instances, the most important initial conservation action will have to occur outside of a given COA, yet the intent of those actions will be to conserve the integrity of the streams within that particular COA. All of the team members agreed that specific attention to, and more intensive conservation efforts within, these 11 COAs will provide an efficient and effective strategy for the long-term maintenance of relatively high quality examples of the various ecosystem and community types that exist within the Meramec River watershed.

In addition to devising the conservation strategy for identifying and mapping COAs, the team also identified other information that needed to be documented during the conservation planning process. This information was captured within a database that can be spatially related to the resulting GIS coverage of the COAs. Specifically, each COA is given a name that generally corresponds with the name of the largest tributary stream, and then each of the following items was documented:

- all of the agencies or organizations that own stream segments within the COA and own portions of the overall watershed or upstream riparian area,
- the specific details of why each AES and VST complex was selected,
- any uncertainties pertaining to the selection of the AES or VST complex and if there are any alternative selections that should be further investigated,
- how these uncertainties might be overcome, such as conducting field sampling to evaluate the accuracy of the predictive models or doing site visits to determine the relative influence of a particular human stressor,
- all of the management concerns within each COA and the overall watershed,
- any critical structural features, functional processes, or natural disturbances,
- what fish, mussel, and crayfish species exist within the COA for each stream size class, and
- any potential opportunities for cooperative management or working in conjunction with existing conservation efforts

All of this information is critical to the remaining logistical aspects of conservation planning that must be addressed once geographic priorities have been established.

Once the core team finalized the conservation strategy and had completed the conservation plan for the pilot area, the state was partitioned into four “regions” with each of these regions containing four EDUs. Regional teams of aquatic resource professionals were then established for each region. Each team consisted of six or more resource managers/biologists with detailed and extensive knowledge of the stream resources within the region they were assigned. Three-day conservation planning sessions were held in each region during summer and early fall of 2004. During these three-day sessions, the regional team used the conservation strategy to develop conservation plans for each of the EDUs within their region.

Conservation plans have been completed for all 17 EDUs in Missouri. Statewide, a total of 158 COAs were identified through the above assessment and planning process (Figure 8.2). These COAs represent the broad diversity of stream ecosystems and riverine assemblages within Missouri and cover a relatively small percentage of the landscape. Specifically, the COAs contain 10,915 km of stream, which represents 6.3% of the 174,059 km of stream within Missouri. In terms of land area, the COAs cover 11,331 km² (2.8 million acres), or just 6.6% of the state.

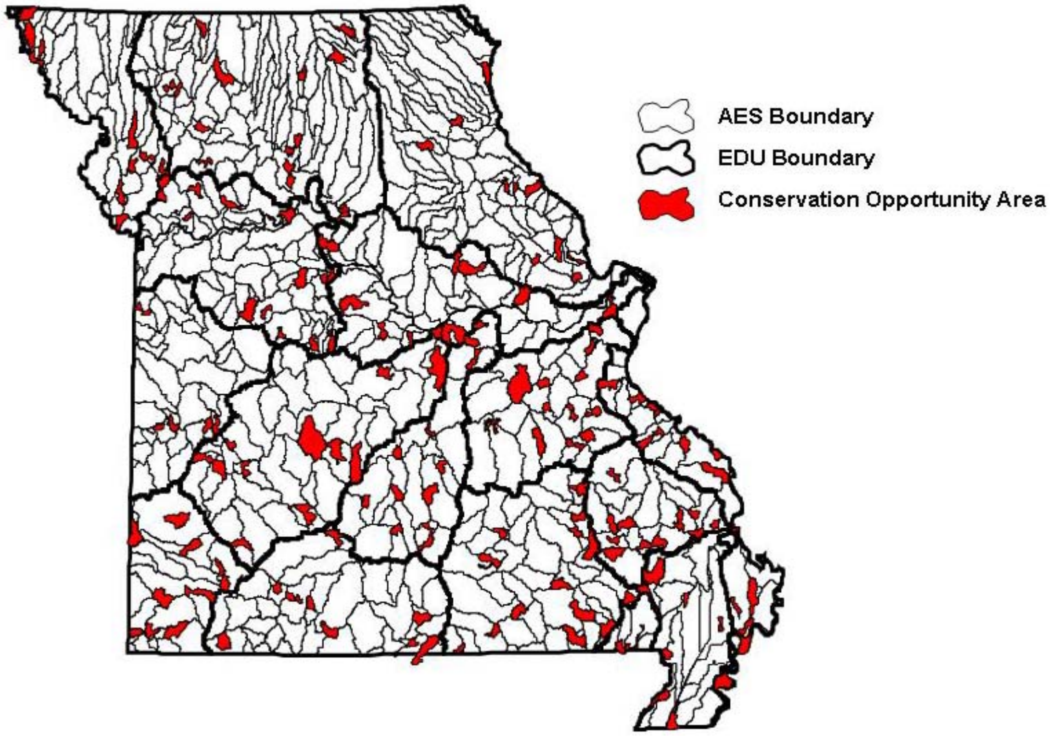


Figure 8.2. Map showing all 158 freshwater Conservation Opportunity Areas that were selected for Missouri. Taking measures to conserve all of these locations represents an efficient approach to representing multiple examples of all the distinct species, stream types, and watershed types that exist within the state.

8.4. Discussion and Limitations

The COAs identified during the statewide conservation planning encompass approximately 6.3% of the total stream miles in the state. Currently, 5% of the total stream miles in Missouri are in public ownership. Consequently, there nearly as many miles currently in public ownership as what the conservation planning results suggest is minimally required to represent the “full range” of variation in stream ecosystem types and multiple populations of all fish, mussel, and crayfish species that occur within the state. However, the results of our gap analyses, presented in Chapter 7, clearly illustrate that the existing network of conservation lands does not even come close to effectively representing the full spectrum of riverine habitats and species that occur in Missouri, especially when more stringent criteria (e.g., connectivity) are used. This irony illustrates the importance of location and spatial arrangement for conserving riverine biodiversity, which heretofore has not been considered in the acquisition of conservation lands.

The 158 COAs that were identified and mapped across the state provide a blueprint for holistic conservation of the freshwater ecosystems within Missouri, as opposed to the largely random and patchwork approach used in the past. These areas can be, and already are being, used to guide protection efforts such as land acquisitions, restoration efforts, and regulatory activities like the permit review process administered under the Clean Water Act. These areas also provide an ideal template for research designed to elucidate fundamental ecological processes within riverine ecosystems since they generally represent the least disturbed examples of the various stream ecosystems that exist within Missouri.

During the conservation planning process we found that the local experts are often humbled by the GIS data. Often, what appear to be the best places to conserve are those places that the local managers know little or nothing about. This exemplifies that the world is a big place, and we cannot expect a handful of experts to know every square inch of an Ecological Drainage Unite (i.e., 10,000+ km^2). At the same time we also found that the GIS data are often insufficient and, if solely relied upon, would often lead to poor decisions. There were several instances where the GIS data identified a particular location, while the local experts quickly pointed out that, for example, the sewage treatment facility just upstream had one of the worst spill records in the state, and fish kills occur almost on an annual basis. While the GIS data show the location of the sewage treatment facility, they do not contain this more detailed information. Obtaining and capture this type of information within a GIS must become a priority.

We were pleasantly surprised by the fact that even in the most highly altered and severely degraded landscapes we were able to identify “hidden jewels” that have somehow escaped the massive landscape transformations and other insults in neighboring watersheds. This experience revealed the social aspects of land use patterns described by Meyer (1995). Yet, in many instances these relatively high quality locations were quite small and therefore highly susceptible to any future changes in local or watershed conditions. Those locations facing any potential immediate threats must be identified and the necessary conservation actions must be put into action quickly, otherwise these “hidden jewels” could be lost forever.

The conservation strategy we developed initially focused on representing all of the distinct watershed (AES-Types) and stream types (Valley Segment Types) within each EDU. In every instance, this initial strategy of ensuring the representation of these abiotic targets, we successfully represented 95-100% of the biotic targets within the initial set of COAs. This is especially surprising in the Ozark Aquatic Subregion, which contains numerous local endemics with very restricted and patchy distributions. These results suggest that our classification units do a good job of capturing the range of variation in stream and watershed characteristics that are partly responsible for the patchy distribution of these species. These results also illustrate the utility of abiotic targets for freshwater conservation planning, which can prove critical for regions lacking sufficient biological data.

Another surprising result was that we were able to represent all of the abiotic and biotic targets within a relatively small fraction of the overall resource base (~6%). Unfortunately, the area that must be managed in order to protect/restore the ecological integrity of any given COA is often substantially larger and much more daunting than the boundaries we delineated. However, the spatially-explicit nature of the COAs provides focus for resource managers, because even when on-the-ground management is far removed from one of these priority locations, the streams and assemblages within each COA are the ultimate focus of conservation action.

When we began our project we recognized the fact that, whenever possible, priorities should be established at a scale that managers can understand and use (e.g., individual stream segments) in order to apply spatially-explicit conservation actions. Each team of local experts found the conservation planning process much more useful than previous planning efforts they were involved in, which identified relatively large areas as priorities for conservation. The managers stated that, because we selected localized complexes of specific stream segments, much of the guesswork on where conservation action should be focused has been taken “out of the equation,” which will expedite conservation action.

Since conservation efforts cannot be initiated immediately within all of the COAs, priorities must be established among the COAs in order to develop a schedule of conservation action (Margules and Pressey 2000). For Missouri, this will initially take place within each EDU and then again from a statewide perspective. An important aspect of generating a “comprehensive” plan is that conservation is often driven by opportunity, and by identifying a portfolio of priority locations quick action can be taken when opportunities arise (Noss et al. 2002).

Selecting COAs is the first step toward effective biodiversity conservation, and the Gap Analysis Program is providing data critical to this task. Yet, establishing geographic priorities is only one of the many steps in the overall process of achieving real conservation. Achieving the ultimate goal of conserving biodiversity will require vigilance on the part of all responsible parties, with particular attention to addressing and coordinating the many remaining logistical exercises.

CHAPTER 9

Summary, Conclusions, and Management Implications

When we save a river, we save a major part of an ecosystem, and we save ourselves as well because of our dependence--physical, economic, spiritual,--on the water and its community of life.

Tim Palmer

Biodiversity refers to the variety and variability among living organisms and the environments in which they occur and is recognized at genetic, population, species, community, ecosystem, and landscape levels of organization (U.S. Congress 1987, Noss 1990). The goal of biodiversity conservation is to reverse the processes of biotic impoverishment at each of these levels of organization. Ecological and evolutionary processes ultimately are as much a concern in a biodiversity conservation strategy as are species diversity and composition. Thus, biodiversity conservation represents a significant step beyond endangered species conservation (Noss 1991, Scott et al. 1991).

While much attention has been focused on the global losses of terrestrial biodiversity, especially in tropical ecosystems, comparatively little attention has been given to the alarming declines in freshwater biodiversity. Similarly, while GAP has made enormous strides in developing and conducting coarse-filter biodiversity assessments for terrestrial ecosystems, much less has been accomplished for aquatic ecosystems. Presently, however, state or regional aquatic GAP projects have been initiated or completed in over twenty states and some of these projects are forging new territory by focusing on lake and marine/estuarine ecosystems, which present new theoretical and technological challenges.

The principal objective of the Missouri Aquatic GAP Project was to identify riverine ecosystems and native species not adequately represented (i.e., gaps) within the existing matrix of conservation lands. In addition, we wanted to provide spatially explicit data that could be used by natural resource professionals, legislators, and the public to make more informed decisions for prioritizing opportunities to fill these conservation gaps and to devise strategic approaches for developing effective long-term strategies for conserving freshwater biodiversity. To accomplish these objectives we developed and compiled four types of geospatial information; 1) maps of relatively distinct riverine ecosystems defined at multiple spatial scales; 2) predicted distribution maps for all 315 fish, mussel, and crayfish species; 3) local, watershed, and upstream network public ownership and stewardship statistics for every stream reach; and 4) maps of various human stressors that affect the ecological integrity of riverine ecosystems and a composite Human Stressor Index for each Aquatic Ecological System.
Our gap analyses quantified the representation of both abiotic and biotic elements of biodiversity. Analyses for the abiotic elements assessed how well the various stream types (Valley Segment Types) and watershed types (Aquatic Ecological System Types)

are represented within the existing matrix of public lands set aside for long term maintenance of biodiversity. These analyses attempt to assess the representation of the distinct riverine habitats/ecosystems across the Missouri landscape, which may prove more useful than assessing representation of individual species (Angermeier and Schlosser 1995). Our analyses for the biotic elements (fish, mussel and crayfish species) follow those used in previous GAP projects dealing with terrestrial vertebrates. However, our statistics are presented in terms of length, not area, since we are dealing with linear and not polygonal data. Furthermore, we also examined the number of distinct locations in which each species is represented in status 1 or 2 lands in order to assess redundancy in representation of distinct population subunits.

The conservation status statistics for each biodiversity element were examined from a statewide perspective and then further examined within an ecosystem context. Specifically, we examined the representation of each biodiversity element within the context of our Aquatic Subregions and Ecological Drainage Units. We believe that these analyses provide a more holistic ecosystem assessment of representation than the statewide analyses.

At first glance, from a statewide perspective, the results of our gap analyses appear somewhat encouraging and also surprising when you consider that only about 5% of the stream miles within Missouri are contained within public lands and less than 1% flow through status 1 or 2 lands. Despite these low percentages, a relatively high percentage of the distinct stream types (Valley Segment Types) (77%) and native species (85%) are represented in lands set aside for the long-term maintenance of biodiversity and in many instances these biodiversity elements are represented within a significant length of stream (e.g., >50 km).

Most of the 17 stream types that are not represented in status 1 or 2 lands are smaller headwater or creeks, which contain relatively few species compared with larger streams. However, headwater streams contain distinct freshwater assemblages, represent the maximum interface between the terrestrial and aquatic environment, and have a significant influence on downstream processes and biotic communities (Karr and Schlosser 1978). It has been suggested that conservation efforts in headwater regions should provide substantial, multiple benefits to adjacent and downstream areas (Karr and Schlosser 1978; Johnston et al. 1990; Haycock et al. 1993). Yet, as we discuss below, filling these conservation gaps must be carried out within a broader perspective that focuses on representing complexes of distinct stream types, paying particular attention to representing a range of stream sizes.

Forty five species (32 fish, 5 mussels, and 8 crayfish) were identified as having none of their predicted distribution contained within status 1 or 2 lands. An additional 24 species were identified as having less than two distinct population subunits represented in status 1 or 2 lands. Most of these 69 species are listed as either state or globally rare, threatened or endangered. The highest concentration of these “underrepresented species” occurs mainly within three areas of the state; 1) the Mississippi River below the confluence with the Missouri River, 2) the Mississippi Alluvial Basin (MAB) Aquatic

Subregion, and c) the Neosho Ecological Drainage Unit (EDU), which is located in southwestern corner of Missouri. The fact that most of these species are either large-river species or local endemics, presents a significant challenge to stream resource managers. Conserving large rivers is obviously difficult due to the enormous land area that must be managed, but also the diversity and cumulative nature of the human disturbances adds to the complexity of the management efforts (Galat and Lipkin 2000). Local endemics present a management challenge because, as our extensive life-history literature reviews for this project showed, very little is known about the life-history requirements of these species. Furthermore, in Missouri, these local endemics occur as widely scattered populations across the state, which places a strain on the limited financial and human resources dedicated to freshwater biodiversity conservation.

The distinct species and unique assemblages found within the MAB and the Neosho EDU (Pflieger 1971; 1989), certainly warrant more attention from resource management agencies in Missouri. The extremely limited amount of public land within these two areas of the state requires that private land conservation initiatives play a prominent role in the long-term conservation of these unique assemblages. However, measures to secure at least some of the lands within the Conservation Opportunity Areas, identified for these regions (see Chapter 8), should also be a high priority.

Some of the most illuminating results from our analyses were revealed by our analyses that assessed the representation of Aquatic Ecological Systems (AESs). For these analyses we used a hierarchical set of criteria. Using the least stringent critieria, which simply required that all stream sizes within an AES to be represented in status 1 or 2 lands, only 19 of the 542 individual AESs were determined to be “effectively represented” and these 19 only represented 6 of the 39 distinct AES-Types. Applying more stringent criteria, which required representation of the various stream sizes as an interconnected matrix, only 12 of the 19 met the criteria. Finally, using all available human stressor data only 4 of the remaining 12 were considered to have relatively high ecological integrity and all four of these AESs were classified as the same AES-Type (Jack’s Fork). These results collectively illustrated the fragmented nature of public lands in Missouri, particularly as they apply to the conservation of riverine ecosystems, and also the level of human disturbance facing those streams within the existing matrix of conservation lands. Clearly, much more emphasis must be placed on the spatial arrangement of future conservation lands, similar to the strategies used in the statewide conservation planning exercises outlined in Chapter 8. These results also illustrated the fact that representation within the matrix of existing conservation lands does not ensure effective long-term conservation for riverine ecosystems. A broader assessment of ecological integrity must be carried out in order to more accurately assess conservation gaps. However, there is certainly a need for much more research addressing how to quantify the degree of human disturbance affecting any particular stream or watershed.

The representation of both abiotic and biotic elements dropped considerably when the analyses were conducted separately for each Aquatic Subregion and EDU (i.e., within an ecosystem context). These analyses clearly revealed the enormity of the challenge we face when it comes to the long-term conservation of freshwater biodiversity in

Missouri. More specifically, if you agree with the contentions of the stream resource managers in Missouri (see Chapter 8) that measures should be taken to holistically conserve each EDU in the state, then our EDU-level analyses provide direct insight into how well this conservation objective is being met. The clearest perspective on how well we are achieving this objective is provided by a specifically examining the “best case scenario” in terms representation of abiotic and biotic elements, which is the Black/Current EDU within the Ozarks. Twenty of the 54 VSTs (37%) and 32 of the 187 native species (17%) that occur within this EDU are not represented in status 1 or 2 lands. Furthermore, only one of the 9 AES-Types that occur within this EDU have all stream sizes represented either separately or as an interconnected matrix. Again, these statistics represent the best case scenario, whereas in many other EDUs, like the Neosho or St. John’s Bayou, we are essentially starting with a “clean slate” in terms of representation. These results clearly indicate that the existing public lands in Missouri do not even come close to holistically representing the full spectrum of freshwater biodiversity, especially at higher levels of ecological organization.

Since most of Missouri and its stream resources are within private ownership, successful conservation of freshwater biodiversity will require creative partnerships between resource agencies and private land owners. The many federal and state conservation incentive programs that are currently used as management tools are certainly a step in the right direction. However, we believe the results our gap analyses illustrate the need for a more strategic approach to where these conservation measures applied on the landscape. Randomly applying the conservation measures across the landscape will likely not provide the same level of benefits as would efforts directed at restoring and protecting key locations across the riverscape that represent the diversity of freshwater ecosystems in Missouri. The data we have developed for the Missouri Aquatic GAP Project are perfectly suited to develop such strategies as was illustrated in the statewide conservation plan discussed in Chapter 8. However, selecting Conservation Opportunity Areas is just the first step toward effective biodiversity conservation. Achieving the ultimate goal of conserving biodiversity will require vigilance on the part of all responsible parties, with particular attention to addressing and coordinating any remaining logistical tasks.

CHAPTER 10

Product Use and Availability

10.1 How to Obtain the Products

It is the goal of the Gap Analysis Program and the USGS Biological Resources Division (BRD) to make the data and associated information as widely available as possible. Use of the data requires specialized software called geographic information systems (GIS) and substantial computing power. Additional information on how to use the data or obtain GIS services is provided below and on the GAP home page (URL below). While a CD-ROM of the data will be the most convenient way to obtain the data, it may also be downloaded via the Internet from the national GAP home page at:

http://www.gap.uidaho.edu/

The home page will also provide, over the long term, the status of Missouri's Aquatic GAP-related projects, future updates, data availability, and contacts. Within a few months of this project's completion, CD-ROMs of the final report and data should be available at a nominal cost—the above home page will provide ordering information. To find information on Missouri Aquatic GAP's project status and data, follow the links to "project information" and then to Missouri.

10.2 Minimum GIS Required for Data Use

The data for the Missouri Aquatic Gap Project were developed using ArcView 3.x and ArcGIS 8.x and 9.x (ArcInfo, ArcMap and ArcCatalog). The minimum GIS tool required to read and work with the data is ArcView, preferably version 3.3.

These are large data layers and will require several gigabytes of hard disk space if you want to load all of the information on your computer at once. Obviously, the faster the processor, and the more hard disk space and RAM you have, the better off you will be. However, these data can be successfully used on machines that meet the minimum hardware requirements for ESRI software.

10.3 Disclaimer

Following is the official Biological Resources Division (BRD) disclaimer as of 29 January 1996, followed by additional disclaimers from GAP. Prior to using the data, you should Although these data have been processed successfully on a computer system at the BRD, no warranty expressed or implied is made regarding the accuracy or utility of the data on any other system or for general or scientific purposes, nor shall the act of distribution constitute any such warranty. This disclaimer applies to both individual use of the data and aggregate use with other data. It is strongly recommended that these data are directly acquired from a BRD server [see above for approved data providers] and not indirectly through other sources which may have changed the data in some way. It is also strongly recommended that careful attention be paid to the content of the metadata file associated with these data. The Biological Resources Division shall not be held liable for improper or incorrect use of the data described and/or contained herein. These data were compiled with regard to the following standards. Please be aware of the limitations of the data. These data are meant to be used at a scale of 1:100,000 or smaller (such as 1:250,000 or 1:500,000) for assessing the conservation status of animals and vegetation types over large geographic regions. The data may or may not have been assessed for statistical accuracy. Data evaluation and improvement may be ongoing. The Biological Resources Division makes no claim as to the data's suitability for other purposes. This writable data may have been altered from the original product if not obtained from a designated data distributor identified above.

10.4 Metadata

Proper documentation of information sources and processes used to assemble GAP data layers is central to the successful application of GAP data. Metadata documents the legacy of the data for new users. The Federal Geographic Data Committee (1994, 1995) has published standards for metadata and NBII (http://www.nbii.gov) has updated those standards to include biological profiles. Executive Order 12906 requires that any spatial data sets generated with federal dollars will have FGDC-compliant metadata.

Remember, metadata describes the development of the spatial data set being documented. If there are companion files to the GIS data, use metadata to reference (reports, spreadsheet, another GIS layer).

10.5 Appropriate and Inappropriate Use of these Data

All information is created with a specific end use or uses in mind. This is especially true for GIS data, which is expensive to produce and must be directed to meet the immediate program needs. Therefore, we list below both appropriate and inappropriate uses. This list is in no way exhaustive but should serve as a guide to assess whether a proposed use can or cannot be supported by GAP data. For most uses, it is unlikely that GAP will provide the only data needed, and for uses with a regulatory outcome, field surveys should verify the result. In the end, it will be the responsibility of each data user to determine if GAP data can answer the question being asked, and if they are the best tool to answer that question.

Scale: First we must address the issue of appropriate scale to which these data may be applied. The data were produced with an intended application at the ecoregion level, that is, geographic areas from several hundred thousand to millions of hectares in size.

The data provide a coarse-filter approach to analysis, meaning that not every occurrence of every plant community or animal species habitat is mapped, only larger, more generalized distributions. The data are also based on the USGS 1:100,000 scale of mapping in both detail and precision. When determining whether to apply GAP data to a particular use, there are two primary questions: do you want to use the data as a map for the particular geographic area, or do you wish to use the data to provide context for a particular area?

Appropriate Uses:
1. Statewide biodiversity planning
2. Regional (Councils of Government) planning
3. Regional habitat conservation planning
4. County comprehensive planning
5. Large-area resource management planning
6. Coarse-filter evaluation of potential impacts or benefits of major projects or plan initiatives on biodiversity, such as utility or transportation corridors, wilderness proposals, regional open space and recreation proposals, etc.
7. Determining relative amounts of management responsibility for specific biological resources among land stewards to facilitate cooperative management and planning.
8. Basic research on regional distributions of species and to help target both specific species and geographic areas for needed research.
9. Environmental impact assessment for large projects or military activities.
10. Estimation of potential economic impacts from loss of biological resource-based activities.
11. Education at all levels and for both students and citizens.

Inappropriate Uses:
It is far easier to identify appropriate uses than inappropriate ones, however, there is a "fuzzy line" that is eventually crossed when the differences in resolution of the data, size of geographic area being analyzed, and precision of the answer required for the question are no longer compatible. Examples include:

1. Combining GAP data with other data finer than 1:100,000 scale to produce new hybrid maps or answer queries.
2. Generating specific areal measurements from the data finer than the nearest thousand hectares (minimum mapping unit size and accuracy affect this precision).
3. Establishing exact boundaries for regulation or acquisition.
4. Precisely quantifying the abundance, health, or condition of any feature.
5. Establishing a measure of accuracy of any other data by comparison with GAP data.
6. Altering the data in any way and redistributing them as a GAP data product.
7. Using the data without acquiring and reviewing the metadata and this report.

CHAPTER 11

Training, Publications, and Presentations

Training Sessions

MoRAP has held nine training workshops in order to provide training to individuals interested in implementing our methods in their respective states (Table 1). Specifically, personnel from the following state and federal agencies and academic institutions have participated in these training workshops;

Federal Agencies:
U.S. Department of Defense, U.S. Environmental Protection Agency, U.S. Forest Service, U.S. Fish and Wildlife Service, and U.S. Geological Survey

State Agencies:
Arkansas Game and Fish Commission, Colorado Division of Wildlife, Florida Fish and Wildlife Commission, Illinois Department of Natural Resources, Maine Department of Environmental Protection, Maine Natural Heritage Program, Maryland Department of Natural Resources, Michigan Department of Natural Resources, Missouri Department of Conservation, Missouri Natural Heritage Program, Virginia Department of Game & Inland Fisheries, and Wisconsin Department of Natural Resources

Academic Institutions:
Kansas State University, Iowa State University, Ohio State University, South Dakota State University, University of Georgia, University of Illinois, University of Maine, University of Michigan, University of Nebraska, University of Wyoming, and Virginia Polytechnic and State University

These training workshops have led to the implementation of state or regional aquatic gap projects within the following states: Alabama, Colorado, Florida, Georgia, Hawaii, Illinois, Iowa, Kansas, Maine, Michigan, Minnesota, Montana, Nebraska, New York, Ohio, South Dakota, Virginia, Washington, Wisconsin, and Wyoming.

Overviews and progress reports on these projects can be found on the GAP website at: http://www.gap.uidaho.edu/projects/aquatic/default.htm

Table 11.1 Summary of the training sessions put on by staff at the Missouri Resource Assessment Partnership from 1999-2003.

Dates of Training	Location	Participants	Agency
March 8-10, 1999	Columbia, MO	Jeff Quinn	Arkansas Game and Fish Commission
		Tracy Ford	Arkansas Game and Fish Commission
		Brian Wagner	Arkansas Game and Fish Commission
		Donald Schrupp	Colorado Division of Wildlife
		Billy Schweiger	EPA Region 7
		Ted Hoehn	Florida Fish and Wildlife Commission
		Randy Kautz	Florida Fish and Wildlife Commission
		Liz Kramer	University of Georgia
		Kevin Kane	Iowa State University
		Kelly Arbuckle	Iowa State University
		Dave Day	Illinois Dept. of Natural Resources
		Forrest Clark	U.S. Fish and Wildlife Service
		Dana Limpert	Maryland Department of Natural Resources
		Sharon Sanborn	U.S. DoD, Fort Leonardwood, MO
		Ralph Haeffner	U.S. Geological Survey-Water Resource Division
		John Tertuliani	U.S. Geological Survey-Water Resource Division
		Chuck Berry	South Dakota State University
		Bob Greenlee	Virginia Department of Game & Inland Fisheries
		Leslie Orzetti	U.S. DoD, Legacy Program
Feb 24-25, 2000	Columbia, MO	Steve Wall	South Dakota State University
		Chad Kopplin	South Dakota State University
Aug 28-29, 2000	Orono, ME	Cindy Loftin	University of Maine
		Dave Courtemanch	Maine Dept. of Environmental Protection
		Dan Coker	Maine Natural Areas Program
Oct 29-30, 2001	Columbia, MO	Jim Peterson	GA Cooperative Fish and Wildlife Research Unit
		1 Graduate student	GA Cooperative Fish and Wildlife Research Unit
Nov 13-14, 2001	Columbia, MO	Robin McNeely	Iowa State University
		Patrick Brown	Iowa State University
Feb 8-9, 2002	Columbia, MO	Keith Gido	Kansas State University
		2 Graduate students	Kansas State University
April 1-2, 2002	Columbia, MO	Ann Hogan	Illinois Dept. of Natural Resources
		Chad Dolan	Illinois Dept. of Natural Resources
Aug 8-9, 2002	Columbia, MO	Geoff Henebry	University of Nebraska
		1 Graduate student	University of Nebraska
Oct 29-30, 2002	Columbia, MO	Jana Stewart	U.S. Geological Survey-Wisconsin
		Alex Covert	U.S. Geological Survey-Ohio
		Stephanie Kula	U.S. Geological Survey-Ohio
		Donna Meyers	U.S. Geological Survey-Ohio
		Allain Rasolofoson	U.S. Geological Survey-Michigan
		Kurt Kowalski	U.S. Geological Survey-Michigan
		Steve Achele	U.S. Geological Survey-Michigan
Oct 29-30, 2002	Columbia, MO	Ed Bissell	U.S. Geological Survey-Michigan
		Jim McKenna	U.S. Geological Survey-New York
		Dora Passino-Reader	U.S. Geological Survey-New York
		Kirk Lohman	U.S. Geological Survey-Minnesota, Illinois
		Daniel Fitzpatrick	U.S. Geological Survey-Minnesota, Illinois

Table 11.1 Continued.

Dates of Training	Location	Participants	Agency
		Chris Smith	Wisconsin Dept. of Natural Resources
		Lizhu Wang	Wisconsin Dept. of Natural Resources
		Paul Seelbach	Michigan Dept. of Natural Resources
July 16-17, 2003	Denver, CO	Don Schrupp	Colorado Division of Wildlife
		Shannon Albeke	Colorado Division of Wildlife
		Nathan Nibbelink	University of Wyoming
		Douglas Beard	U.S. Geological Survey-GAP, NBII

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Glossary of Acronyms

AES – Aquatic Ecological System
AML – Arc Macro Language
BRD – Biological Resources Division of USGS
BLM – Bureau of Land Management
CART or C&RT - Classification and Regression Trees
CCC - Cubic Clustering Criterion
CHAID - Chi-squared Automatic Interaction Detector
C&RT or CART - Classification and Regression Trees
COA – Conservation Opportunity Area
CP – Central Plains aquatic subregion
CWCS - Comprehensive Wildlife Conservation Strategy
DEM – Digital Elevation Model
DLG – Digital Line Graph
EDU – Ecological Drainage Unit
EMAP – Environmental Monitoring and Assessment Program
EPA – Environmental Protection Agency
ESRI – Environmental Systems Research Institute
FGDC – Federal Geographic Data Committee, USGS
FLIR - Forward Looking Infrared Radar imagery
GAP – Gap Analysis Program
GAP1 – Gap Analysis Program land management Status 1
GAP2 - Gap Analysis Program land management Status 2
GAP3 - Gap Analysis Program land management Status 3
GAP4 - Gap Analysis Program land management Status 4
GIS – Geographic Information System
GPS – Global Positioning System
Grank – A numeric rank of relative endangerment based primarily on the number of global occurrences of the species.

- G1 – Critically imperiled globally because of extreme rarity or because of some factor(s) making it especially vulnerable to extinction.
- G2 – Imperiled globally because of rarity or because of some factor(s) making it very vulnerable to extinction throughout its range.
- G3 – Either very rare and local throughout its range or found locally (even abundantly at some of its locations) in a restricted range or because of other factors making it vulnerable to extinction throughout its range.
- G4 – Widespread, abundant, and apparently secure globally, though it may be quite rare in parts of its range, especially at the periphery.
- G5 – Demonstrably widespread, abundant, and secure globally, though it may be quite rare in parts of its range, especially at the periphery.
- G? – Unranked: species is not yet ranked globally.
- GX – Extinct: Believed to be extinct throughout range with virtually no likelihood that it will be rediscovered.

HIS – Human Stressor Index

HU – Hydrologic Unit
LTA – Land Type Association
MAB – Mississippi Alluvial Basin aquatic subregion
MDC – Missouri Department of Conservation
Missouri DNR – Missouri Department of Natural Resources
MoRAP – Missouri Resource Assessment Partnership, University of Missouri
MRPP - Multi-Response Permutation Procedure
NHD – National Hydrography Dataset
NHP – Natural Heritage Program
NMS - Nonmetric Multidimensional Scaling
NRCS – Natural Resources Conservation Service
OZ – Ozark aquatic subregion
PCA - Principal Components Analysis
QUEST - Quick, Unbiased, Efficient Statistical Tree
Srank – A numeric rank of relative endangerment based primarily on the number of occurrences of the species within the state.

- S1 – Critically imperiled in the state because of extreme rarity or because of some factor(s) making it especially vulnerable to extirpation from the state.
- S2 – Imperiled in the state because of rarity or because of some factor(s) making it very vulnerable to extirpation from the state.
- S3 – Rare and uncommon in the state.
- S4 – Widespread, abundant, and apparently secure in state, with many occurrences, but the species is of long-term concern.
- S5 – Demonstrably widespread, abundant, and secure in the state, and essentially ineradicable under present conditions.
- S? – Unranked: Species is not yet ranked in the state.
- SX – Extirpated: Species is believed to be extirpated from the state.

SSURGO – Soil Survey Geographic Database
STATSGO – State Soil Geographic Database
SWG – State Wildlife Grants program
TNC – The Nature Conservancy
USDA – United States Department of Agriculture
USFS – USDA Forest Service
USGS – United States Geological Survey
UTM – Universal Transverse Mercator
VST – Valley Segment Type